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PREFACE

Volume 106 contains five outstanding reviews that focus on the impacts of climate change on the water cycle, crop production, and grassland fire management, phosphorus mobility and its effects on water quality, organic matter structure and interactions in the environment, and strategies for more effectively communicating soil science research findings. Chapter 1 discusses the role of drying and rewetting, and freezing and thawing on the mobilization of P to surface waters. Chapter 2 provides strategies for enhancing crop production in water-limited environments. Chapter 3 provides a contemporary and historical view of organic soil matter chemistry and its importance in the environment. Chapter 4 is a timely review on ways to enhance the dissemination of soil science research to the public and the policymakers. Chapter 5 covers the effects of climate change and increases in temperature on fire management of grasslands.

I am grateful to the authors for their fine reviews.

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CHAPTER ONE

Phosphorus Solubilization and Potential Transfer to Surface Waters from the Soil Microbial Biomass Following Drying–Rewetting and Freezing–Thawing

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Abstract

Drying–rewetting and freezing–thawing are two of the most common forms of abiotic perturbations experienced by soils, and can result in the solubilization of phosphorus (P). There is increasing interest in one particular component of soil P that may be especially susceptible to such stresses: the soil microbial biomass. We examine the evidence for the soil microbial biomass acting as a significant source of P in soils and surface waters by studying the literature on the processes responsible for its solubilization and transfer, resulting from abiotic perturbations. These perturbations have been shown to kill up to *circa* 70% of the total microbial biomass in some soils, and in some cases nearly all the additional P solubilized has been attributed to the microbial biomass. The degree to which the soil microbial biomass is affected by abiotic perturbations is highly dependent upon many variables, not the least degree, duration, and temporal patterns of stress, as well as the soil type. It is hypothesized that while abiotic perturbations can solubilize large quantities of P from the soil microbial biomass in some soils, only a small proportion is likely to find its way from the soil to surface waters. This is not to say that this small proportion is not significant with regard to surface water quality and nutrient loss from the soil, and may become more prevalent under future climatic change. We conclude that it is likely that only extreme conditions will elicit large responses with regard to the solubilization and transfer of phosphorus to surface waters.

1. Introduction

Soils experience a wide range of abiotic perturbations, including flooding, heating, freezing, drying, thawing, and compaction. However, drying–rewetting and freezing–thawing are two of the most common forms of abiotic perturbations (Soulides and Allison, 1961), and they play a critical role in soil nutrient dynamics and biogeochemistry. Both these perturbations are predicted to become more extreme with the United Nations Framework Convention on Climate Change (UNFCCC, 2008) stating that "...it is very likely that hot extremes, heat waves and heavy precipitation events will continue to become more frequent..." on a global scale, with many regions experiencing "...increased intensity of droughts and floods as a result of climate change." These trends have potentially significant (but as yet not fully known) consequences for soil nutrient dynamics. The spatial extent of these perturbations is wide, with almost all soils experiencing moisture fluctuations to varying degrees and over different temporal periods due to wetting mechanisms, which include precipitation, riverine flooding, tidal flooding and irrigation. Soils that experience freezing and thawing are more limited in extent because they occur mostly at high altitudes and latitudes.

Despite this, approximately 55% of the total land area in the northern hemisphere experiences seasonally frozen ground (Zhang *et al.*, 2003).

Solubilization of nutrients in soils refers to the release of previously bound or fixed nutrients (such as those immobilized in the microbial biomass) into solution. Increases in nutrient solubilization within dried soils following their rewetting are widely reported particularly with regard to carbon (C) and nitrogen (N) (e.g., Denef *et al.*, 2001; Fierer and Schimel, 2002; Franzluebbers *et al.*, 1996; Gordon *et al.*, 2008; Mikha *et al.*, 2005; Van Gestel *et al.*, 1991; Venterink *et al.*, 2004), but less so for phosphorus (P) (Chepkwony *et al.*, 2001; McNeill *et al.*, 1998; Qiu and McComb, 1995; Turner and Haygarth, 2001; Venterink *et al.*, 2004). Similarly, increases in the solubilization of C and N following soil freezing and thawing are widely reported (Freppez *et al.*, 2007; Wang and Bettany, 1993; Williams and Wheatly, 1991), but there has been little research on the effects it has on P solubilization (Freppez *et al.*, 2007; Ron Vaz *et al.*, 1994). Phosphorus is particularly important as not only is it a vital nutrient for DNA, RNA, and energy transfer in all organisms, but also its importance in the eutrophication of freshwater systems is increasingly being recognized (Conley *et al.*, 2009). Consequently, a better understanding of the pools, sources, and processes affecting its cycling is vital if we are to understand how predicted climate change will affect potential loss of soil P to surface waters and how agriculture can optimize nutrient use at this time of growing concern over food security and production efficiency.

1.1. Rationale

The soil has been identified as one of the key diffuse sources of P, but the role of the various pools of P in the soil in this context is relatively unknown. Phosphorus can exist in many forms in soils, but in simple terms, it generally exists in one of the following forms: as part of recalcitrant mineral particles, in either soluble or adsorbed mineral forms such as colloidal P, or as part of soil organic matter. Both organic and inorganic P may be solubilized following drying and rewetting of soils, although organic P generally dominates (Turner and Haygarth, 2000; Turner *et al.*, 2003b). However, the sources of this P are still poorly understood. Some reports consider that the physical disruption of aggregates upon rewetting (Birch and Friend, 1958; Miller *et al.*, 2005; Powlson and Jenkinson, 1976) and subsequent solubilization of particulates and associated P are the key processes involved, while others claim that the source of solubilized P is mainly the lysis of microbial cells (Grierson *et al.*, 1998; Turner and Haygarth, 2000; Turner *et al.*, 2003b) and native soil organic matter (Jenkinson, 1966; Pulleman and Tietema, 1999; Van Gestel *et al.*, 1991). Many consequences

of the drying–rewetting process in soils are similar to those that occur during freezing–thawing cycles (Lehrsch *et al.*, 1991). Freezing can affect aggregate stability (Larson and Allmaras, 1971) resulting in increased mobilization of soil particulates upon thawing, while the expansion of intracellular fluids during freezing can result in cell lysis and release of solutes (Soulides and Allison, 1961). Physicochemical reactions involving organic matter can also occur and are similar to those during drying–rewetting episodes (Giesy and Briese, 1978).

Theoretically, drying and rewetting or freezing and thawing of soils can result in solubilization of P from any of the P pools in soils. However, there is increasing interest in one particular component of soil organic P that may be especially susceptible to such stresses, and hence constitutes a significant source of solubilizable P in many soils: the soil microbial biomass. This is the living component of soil organic matter, comprising 1–3% of the total organic matter in most soils and by definition excludes higher soil animals and plant roots (Dalal, 1998). Essentially, it comprises microorganisms in the form of bacteria, fungi, actinomycetes, and protozoa as well as nematodes. Despite the microbial biomass being a relatively small fraction of total organic matter in most soils (McNeill *et al.* 1998; Turner and Haygarth 2001), the quantities of P immobilized in the microbial biomass are surprisingly large. Brookes *et al.* (1984) estimated that in the UK the biomass P pool size in arable soils is up to 30 kg P ha^{-1} in the plow layer, while for grassland soils, it often exceeds 100 kg P ha^{-1}. Thus the biomass P in both ecosystems can easily exceed plant P uptake, reported by Blake *et al.* (2000) as between about 5 and 30 kg P ha^{-1} yr^{-1} for a range of crops across Europe.

Drying and rewetting has been shown to kill up to ca. 70% of the total microbial biomass in some soils (Blackwell *et al.*, 2009a; Kieft *et al.*, 1987; Van Gestel *et al.*, 1993), and the dead biomass can act as a significant source of solubilized reactive and unreactive P. Following the air-drying of 15 different New Zealand soils, Sparling *et al.* (1985) demonstrated that the microbial biomass can contribute up to 76% of bicarbonate extractable reactive P. Turner and Haygarth (2001) examined 29 lowland pasture soils. They found a positive relationship between the quantities of microbial biomass P in the original soil and the amount of total soluble P they contained after being subjected to drying–rewetting cycles. They attributed the increases in water soluble P to P release from the microbial biomass. The large contribution of the microbial biomass to solubilized P following drying and rewetting was confirmed by Turner *et al.* (2003b) who demonstrated that up to 95% of the unreactive P solubilized following drying and rewetting of pasture soils was attributable to lysed bacterial cells. While P held in the other soil pools, such as native organic matter and colloidal P, is likely to be solubilized by abiotic perturbations, these reports suggest that, perhaps because of the dynamic nature of the soil microbial biomass, it is more vulnerable to abiotic stresses than other pools. Therefore, it could

potentially be a significant source of solubilizable P in many soils. Furthermore, any P originating from the microbial biomass has the potential for transfer from the soil to surface waters (Qiu *et al.*, 2004; Turner and Haygarth, 2001, Turner *et al.*, 2003a), where it may trigger eutrophication events.

1.2. Objective

In the current paper, we examine the evidence for the soil microbial biomass acting as a significant source of P in surface waters by reviewing the literature on the processes responsible for its solubilization and transfer, resulting from drying–rewetting and freezing–thawing processes. This is considered in light of the changes to the abiotic processes affecting microbial P solubilization, mobilization, and transfer as a result of predicted climate change. Usually, the different sources of soil-derived P in surface waters are considered *en masse*. This is because it is generally difficult to differentiate between the different sources of soil-derived P in surface waters, and even soluble P in soil itself. However, the focus here is on the release of P directly from microbial cells, either through their mortality or biological regulatory and/or metabolic processes. Waterlogging of soils can also result in microbial release of P, but generally this process is neither temporally nor spatially as extensive in soils as drying–rewetting and freezing–thawing processes and therefore is not considered here. We will highlight key gaps in the literature and discuss the research that is required to improve our knowledge. A summary of the key issues covered and how they are interrelated is shown in Fig. 1. This topic has a wide significance in that a better understanding of the processes associated with microbial P released during drying–rewetting and freezing–thawing events will assist in predicting how climate change will affect soil nutrient status and plant–nutrient interactions, the potential loss of P in leachate, and overall soil function.

2. Factors Affecting Solubilization of Phosphorus from the Soil Microbial Biomass

Decreases of more than two-thirds of the original microbial biomass immediately following drying of soils have been reported (Blackwell *et al.*, 2009a; Bottner 1985; Kieft *et al.*, 1987; van Gestel *et al.*, 1993). Logsdail and Webber (1959) reported that most of the bacteria were eliminated in a soil subjected to a sudden, severe frost, while Soulides and Allison (1961) reported that freezing can decrease soil bacterial numbers by up to a third. The basic processes involved in reducing the microbial populations in soils following these perturbations are well documented.

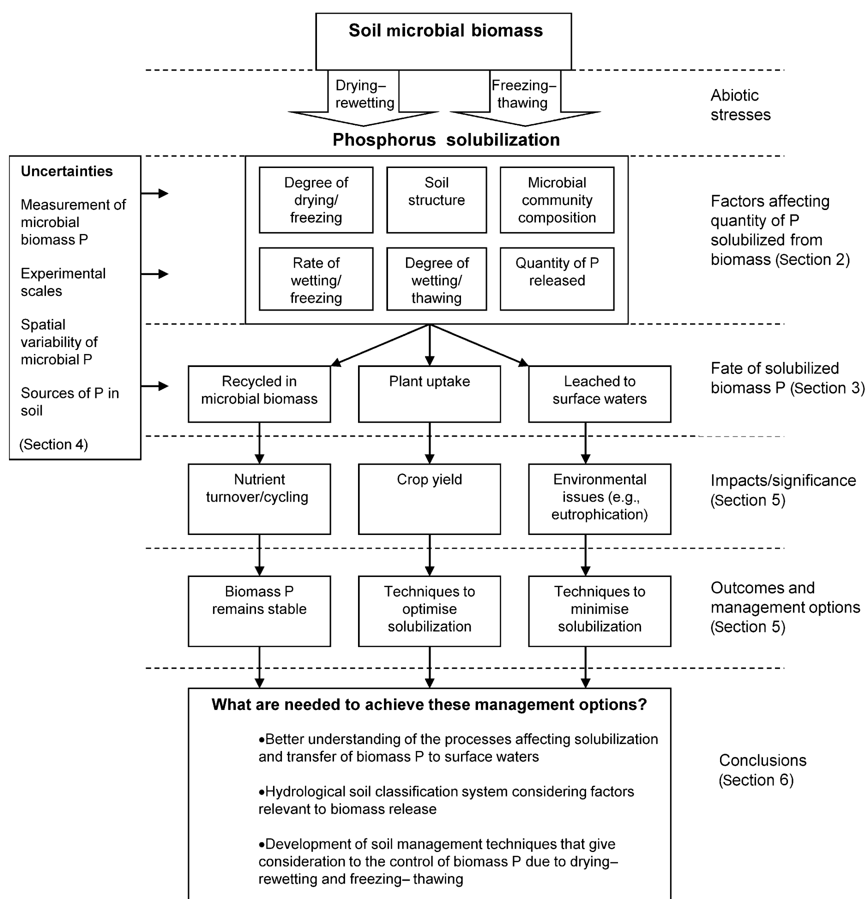


Figure 1 Summary of key factors affecting the solubilization of P from the soil microbial biomass by abiotic processes.

2.1. Primary solubilization processes

At the microbial scale, the main processes by which drying–rewetting and freezing–thawing cycles directly affect soil microorganisms are desiccation, osmotic processes, cell lysis and intracellular fluid expansion upon freezing and cell disruption.

2.1.1. Desiccation, osmotic regulation and osmotic shock

Desiccation, osmotic regulation and osmotic shock occur largely due to the stresses imposed by changes in soil water potential. In simple terms, soil water potential is a measure of the energy required to remove water from

the soil. Soil drying is generally a gradual process during which turgor pressure increases as water is removed from the surrounding soil and so the microbial cells release water (generally thought not to contain P) in order to equilibrate this pressure (homeostatic regulation). This means that mortality and cell lysis do not necessarily occur, although if too extreme, desiccation, and mortality will take place (Halverson *et al*., 2000), resulting in the release of microbial P. Similarly anhydrobiosis occurs in some microorganisms in an attempt to avoid damage during freezing. In this process, bulk water within cells is minimized or even eliminated, leaving behind only miniscule amounts of water forming bound or vicinal water shells around macromolecules, resulting in no free water to form damaging ice crystals (see below), but severe desiccation can occur, again potentially leading to P solubilization (Storey and Storey, 2005). Increases in solute concentrations in the soil solution can occur during drying due to the evaporation of water, while during freezing this increase results from the removal of pure water from the soil solution to form ice crystals in the extracellular spaces, rather than in the cell itself. Both processes set up a steep osmotic gradient across cell membranes causing an outflow of water and consequently cell dehydration, elevated intracellular ionic strength and cell volume reduction, all of which cause stress and potential death. The rewetting of dry soils can cause rapid fluctuations in soil matric potential ranging from -20 MPa to almost 0 (Evans *et al*., 1975), causing a number of different microbial processes to occur. Rapid increases in turgor pressure can occur, resulting in cell damage, plasmolysis and/or death (Harris, 1981; Kieft *et al*., 1987; Salema *et al*., 1982; Storey and Storey, 2005). This results in the release of cell contents of which P is an important component. However, the thicker a microorganisms cell wall, the more likely it is to be able to withstand these pressures (Harris, 1981). Halverson *et al*. (2000) report that, upon rapid rewetting, cells may actively or passively release intracellular organic and inorganic solutes (Christian, 1962; Reed *et al*., 1986), as well as catabolize compatible organic solutes to CO_2 or polymerize solutes to reduce osmotic activity (Berrier *et al*., 1992; Fierer and Schimel, 2003; Reed and Stewart, 1983), thereby avoiding cell lysis and mortality. The chemistry of the rewetting water is also important because if it is not in ionic balance with intracellular solutions this can also increase the likelihood of osmotic shock (Appel, 1998).

2.1.2. Intracellular fluid expansion upon freezing and cell disruption

When ice crystals form they can cause direct physical damage to cells and tissues. Shearing and squeezing stresses can break individual cells trapped among growing ice crystals resulting in cell rupture, and subsequent release of P-containing cytoplasm upon thawing (Storey and Storey, 2005). When ice crystals form inside cells, subcellular architecture can be damaged and

metabolic pathways can be disrupted, again resulting in cell lysis and death (Edwards and Cresser, 1992). However, the degree to which this occurs will depend on the chemical composition of the cytoplasm, as psychrophiles (organisms able to grow at low temperatures) have developed a number of ways of surviving in frozen soils including the production of biomolecules enabling them to function at low temperatures (Madigan *et al.*, 2000; Robinson, 2001; Storey and Storey, 2005).

2.2. Thresholds of solubilization

The degree, rate, and duration of soil drying and rewetting can vary, as can the quantity and chemistry of the rewetting water, all of which can affect microbial responses. Similarly, whether a soil is frozen slowly or rapidly, the minimum temperature, duration of freezing and the number of freeze–thaw events can result in different effects on the soil microbial biomass. All these factors can affect the degree to which the microbial biomass is impacted and the quantity of P released.

2.2.1. Drying–rewetting

The longer a soil remains dry, the less the ability of the biomass to recover upon rewetting, at least in the short term (De Nobili *et al.*, 2006). After 12 years the period of drying was found to make little difference. This has implications for the biomass in regions where droughts can last for several years, and if these become extended, possibly as a result of climate change, biomass populations may become severely depleted. The rate at which a soil dries and rewets affects the stress experienced by organisms, it being both a function of the absolute water potential value and the rate of change of water potential. Slow drying of soil can enable microbial metabolic adjustment which reduces mortality during the drying process (Chao and Alexander, 1984; Roberson and Firestone, 1992). The degree of drying is also important as shown by Kieft *et al.* (1987), who reported greater biomass mortality upon the rewetting of soils dried to -6.9 MPa compared to -2.8 MPa. West *et al.* (1992) found that biomass remained constant until soil moisture dropped to between 0.1 and 0.3 $m^3\ m^{-3}$, but reported that the rates at which soils dry do not affect biomass survival, despite differences in soil texture, organic matter and moisture characteristics. Rate of drying is closely related to the temperature at which drying takes place. Turner and Haygarth (2001) found that while the temperature at which soils were dried did not affect the total quantity of P solubilized, it did affect the rate of solubilization. At higher temperatures the biomass was killed more quickly making microbial P available in soils more rapidly than under drying at lower temperatures. Again this has implications for changes in patterns of drying and rewetting under climate change, with a predicted shift towards higher temperatures. This implies that microbial P will potentially be more

rapidly solubilized, but whether or not it increases the likelihood of leaching will depend upon the nature of rewetting, as discussed in Section 3. The fact that the P is more rapidly available could mean that there is greater opportunity for it to be utilized by other soil organisms or plants, thereby reducing its potential loss.

Soil rewetting usually takes place rapidly following precipitation, irrigation, or flooding as a wetting front penetrates the soil. This can exert some of the greatest stresses experienced by soil microorganisms (Kieft *et al.*, 1987). If the drying phase of a soil is relatively slow, osmotic regulation can take place resulting in only a small change in biomass. Hypothetically, if the rewetting process occurs at a slow rate, then solubilized nutrients from either the biomass or other sources could potentially be taken up by the surviving microbes, stimulating microbial growth, and the biomass would be restored to almost its original status. However, at the microbe scale, slow rewetting is unlikely to occur as most rewetting processes are not subtle enough to constitute slow rewetting. One possible exception is if a soil is rewetted through the introduction of water vapour, which slowly condenses. Therefore, during most rewetting events osmotic shock occurs and some microbes die, resulting in a rapid decrease in biomass and release of microbial P. When rewetting a dried soil at rates ranging from 0 to 48 h, Blackwell *et al.* (2009a) found that the rate of rewetting had no significant effect on the quantity of microbial biomass, thereby supporting this speculation.

The frequency of drying–rewetting cycles is also an important factor in determining the release of biomass P. In experiments where soils were subjected to repeated drying–rewetting cycles, some researchers have reported contrasting results. Almost full recovery of biomass following initial depletions of up to one third the original biomass were found by Bottner (1985) while Butterly *et al.* (2009) found that microbial biomass P and C concentrations were gradually depleted. They both observed a reduction in activity following an initial increase, which they attributed to an initial rapid turnover of microbial nutrients during the first few drying–rewetting cycles, during which the microbial communities adapt to the variable conditions and indeed the community changes (see Section 2.3), such that the effects of drying and rewetting are reduced with regard to cell mortality as more resilient species dominate (Franzluebbers *et al.*, 1994; Lundquist *et al.*, 1999; Mikha *et al.*, 2005). Additionally, the active fraction of the biomass is likely to be depleted, so although no shift in community or biomass may be observed, the change in activity could potentially reflect a change in the inherent properties of the population (van Gestel *et al.*, 1993). In contrast, Miller *et al.* (2005) detected a negligible decrease in activity following repeated rewetting of chaparral soils despite decreases in total soil C, although the frequency of drying and rewetting was identified as an important factor controlling the amount of CO_2 emitted during rewetting episodes.

The translocation of water by roots may also cause localized drying–rewetting within soils and therefore, diffused microsite drying–rewetting cycles may occur, in addition to the larger drying–rewetting cycles associated with precipitation, irrigation, and flooding events. Murray *et al.* (2006) observed reductions in soil microbial biomass in soils which had received N fertilizer. They speculated that this was caused by soil drying due to increased evapotranspiration by the larger plants resulting from fertilizer application. It is likely that most soils will be adapted to these microvariations and it is probably the larger, more extreme events that would result in mobilization and leaching of biomass P.

2.2.2. Freezing and thawing

Soulides and Allison (1961) found that repeated freezing of an air-dried temperate soil did not kill bacterial cells, but that one treatment was sufficient to eliminate most of the organisms that were unable to withstand the impact of freezing. They also reported that, as for drying of soils, the rate of bacterial destruction by freezing was highest when bacteria was in its growth stage and almost negligible during the decline stage. Feng *et al.* (2007) also found that repeated freeze–thaw cycles did not significantly affect bacterial populations, but did cause a gradual decline in fungal biomass, while Sjursen *et al.* (2005) observed that extended periods of freezing did not affect the fungal biomass, but did cause a decline in bacterial populations. The rate at which a soil freezes is also important because it not only affects the size of ice potentially damaging ice crystals, but can also affect the opportunity the microbial biomass has to adapt to the changing conditions. At high freezing rates ($>1.4\ ^{\circ}C\ h^{-1}$) decreases in biomass have been reported, while at relatively lower freezing rates, they remain unaffected (Lipson *et al.*, 2000). This could be because slower freezing rates mean microbes have more opportunity to purposely dehydrate (anhydrobiosis), thereby removing all bulk water which potentially could cause harm during freezing (Storey and Storey, 2005). Similarly during moderate freeze–thaw treatments many soil microbes are able to survive both single and multiple freeze–thaw cycles (Grogan *et al.*, 2004), but not single severe freezing, when temperatures fall below $-10\ ^{\circ}C$ (Mikan *et al.*, 2002; Soulides and Allison, 1961). This is because while bulk soil water freezes below $0\ ^{\circ}C$, soil particles continue to have thin liquid films around them in which microbes not only survive, but also continue activity typically down to $-10\ ^{\circ}C$, but sometimes down to $-40\ ^{\circ}C$ (Romanovsky and Osterkamp, 2000; Schimel and Mikan, 2005). The duration of freezing is also likely to be important, because as the freezing period is extended, soil aggregates will become increasingly dehydrated as water is drawn towards the freezing front. Sometimes a freezing front will move progressively down through a soil profile, with the excluded solutes becoming increasingly concentrated and having implications for osmotic processes (see above). Edwards *et al.* (2006)

reported that in boreal soils a large and rapid (7–10 days) decrease in microbial biomass occurred following thawing of soils maintaining temperatures around 0 °C. They attribute this primarily to competition for nutrients from plants becoming active at these temperatures. Similar observations were made by Larson *et al.* (2007) in subarctic soils. Thus it would appear that in soils accustomed to low temperatures it is extreme freezing events and extended periods of defrosting that are most detrimental to the biomass, while repeated, moderate freeze–thaw cycles have less effect. In contrast, single moderate freezing events are significantly detrimental to the biomass in temperate soils. However, temperature threshold values for specific soils and processes cannot be defined (Matzner and Borken, 2008).

2.3. Impacts on microbial communities

Different groups and species of microorganisms react differently to varying degrees, duration, and frequency of desiccation and freezing. Most microorganisms are capable of surviving periods of desiccation form endospores, cysts, or other resistant structures, whereas *Arthrobacter* and some rod-shaped bacteria can survive drying as more-or-less structurally unmodified cells (Chen and Alexander, 1973). Similar methods are adopted by various microorganisms to survive freezing, with many forming spores that germinate upon thawing, while others dehydrate to avoid the damaging effects of ice crystal formation. Some have developed the ability to produce antifreeze agents, so that they can survive much colder temperatures (Robinson, 2001; Storey and Storey, 2005). The dominant varieties of soil microorganisms are therefore likely to affect the quantity and form of biomass P released. For example, those forming cysts or spores are likely to release both cell wall and cytoplasm P compounds from the parent cell. Figure 2 illustrates a simplified version of the processes that may occur and how P can be mobilized from soil microbes during a drying–rewetting cycle. In Fig. 2A, soil microbes are fully hydrated in a moist soil, and P is present in the cytoplasm and cell walls of intact cells. In Fig. 2B the soil undergoes drying so that desiccation occurs, cells rupture and P is released into the soil solution and cell walls become part of the soil organic matter. In Fig. 2C the soil, undergoes drying, but cells osmoregulate and survive, releasing no microbial P. In Fig. 2D the soil undergoes rewetting, and some of the P released during desiccation is taken up by other and new soil microbes. Some P remains in the soil solution and P in the cell walls of dead microbes is incorporated into the soil organic matter. In Fig. 2E the soil undergoes rapid rewetting, causing osmotic shock and cell lysis. Phosphorus is released into the soil solution and cell walls form part of the soil organic matter. Finally, in Fig. 2F the soil undergoes slow rewetting, cells remain intact and rehydration occurs. This final scenario is unlikely but could possibly occur, as discussed above.

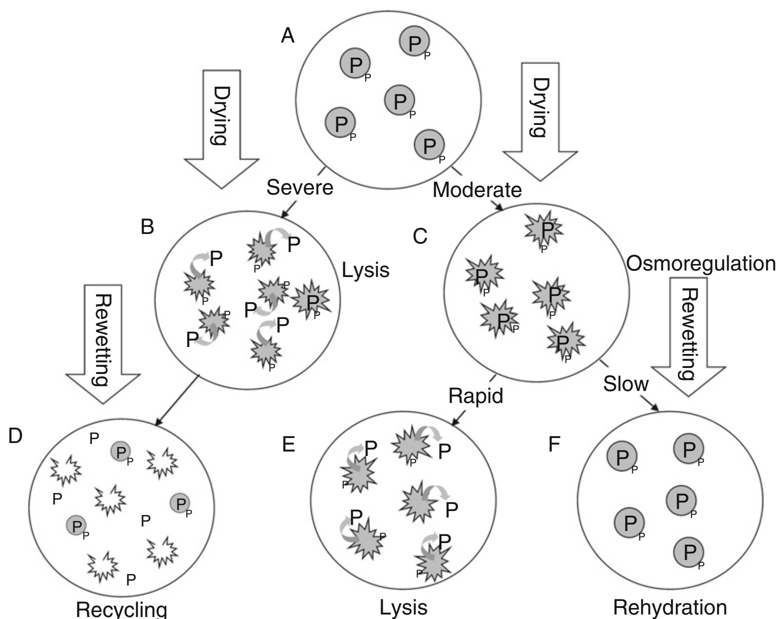


Figure 2 Simple diagram of the effects of drying–rewetting events on the soil microbial biomass and potential for P solubilization.

In addition to the type of microbial populations present, the inherent properties of the biomass, such as physiology and metabolic state, can also affect responses to perturbations. Soulides and Allison (1961) found that cells in a sporulated or resting state are less susceptible to drying or freezing than rapidly growing cells, while Bottner (1985) and van Gestel *et al.* (1993) reported that active biomass is more vulnerable to drying than the dormant fraction.

Following rewetting, different microorganisms may have a reproductive advantage over others depending on various factors. For example, extreme desiccation may favur gram-positive bacteria and fungi, which are more likely to survive such events but are slow-growing, while less extreme events may favur organisms such as gram-negative bacteria. These are able to multiply more rapidly due to the increased availability of nutrients (Bushby and Marshall, 1977; Fierer *et al.*, 2003), but are more susceptible to drying (van Gestel *et al.*, 1993). Fungi are generally reported to be more susceptible to freezing than bacteria (Feng *et al.*, 2007), but this does vary depending upon the temperature, rate, and duration of freezing. Such differences imply that drying–rewetting and freezing–thawing cycles will cause changes in soil microbial communities, but that these changes will to some extent be dependent on the characteristics of drying and freezing. While some

evidence exists to show that community shifts can occur (Feng *et al.*, 2007; Gordon *et al.*, 2008; Lundquist *et al.*, 1999), for example yeasts recolonize the soil more successfully than filamentous fungi and bacteria (Sparling and Cheshire, 1979), it is not always the case. Rather, it depends largely on whether soils regularly experience drying–rewetting or freezing–thawing events and if the microbial community is adapted to such conditions (Fierer *et al.*, 2003; Griffiths *et al.*, 2003; Hamer *et al.*, 2007; Koponen *et al.*, 2006). Additionally, drying–rewetting and freezing–thawing cycles are likely to cause a shift in the inherent properties of the microbial population with a trend towards greater survival of less active, more mature organisms (Bolter *et al.*, 2005; Soulides and Allison, 1961; van Gestel *et al.*, 1993), which will be less susceptible to further perturbations (see above). Figure 3 shows how microbial community changes may occur due to drying and rewetting within a single soil aggregate. Figure 3A shows a moist aggregate with equal numbers of gram-positive and gram-negative bacterial cells and fungi. In Fig. 3B, the aggregate experiences an extreme drying event. All gram-negative bacterial cells die due to their inability to survive extreme drying, and some

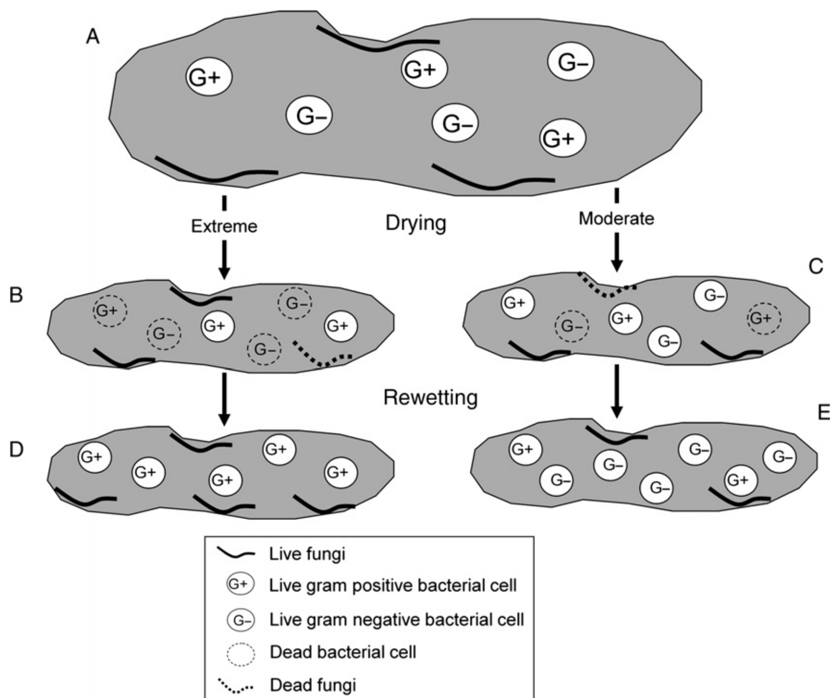


Figure 3 Possible effects of drying–rewetting events on soil microbial communities.

gram-positive bacterial cells and fungi also die. In Fig. 3C, the aggregate experiences a moderate drying event. Some of the three types of micro-organisms die and some of the three types also survive. Figure 3D shows, that upon rewetting, gram-positive bacteria and fungi reproduce using nutrients released by the dead gram-negative cells, and gram-positive bacteria and fungi dominate. Figure 3E shows that upon rewetting, surviving gram-negative bacterial cells are able to regenerate more rapidly than gram-positive bacterial cells and fungi, and consequently gram-negative bacteria dominate.

3. Factors Controlling Transfer to Surface Waters

3.1. Internal cycling of phosphorus

There is substantial evidence indicating that after a soil experiences a perturbation, the rate of microbial turnover and activity increases, probably as a result of the recycling and processing by the surviving biomass of nutrients released from the original biomass and other sources. Many studies have considered the observed pulse of CO_2 emission from soils following rewetting (Birch, 1958; Bottner, 1985; Butterly *et al.*, 2009; Fierer and Schimel, 2003; Mikha *et al.*, 2005; Miller *et al.*, 2005) and thawing (Schimel and Clein, 1996; Soulides and Allison, 1961) as an indicator of microbial activity. The actual processes involved have been topics of great debate. Originally it was thought that physical soil processes could explain these flushes of activity, whereby desiccation and freezing of soil organic matter resulted in exposure to enzymes of previously inaccessible surfaces of organic or organo-mineral colloids due to aggregate fragmentation or increased porosity (Birch, 1958; Soulides and Allison, 1961). Powlson and Jenkinson (1976) confirmed that the physical disruption of the soil structure and substrate desorption from surfaces along with increased microbial mobility, all caused by drying–rewetting cycles, increases the solubilization of organic compounds including organic P. Similar results are reported by Dergacheva and Dedkov (1977) and Giesy and Briese (1978) following freezing–thawing of soils. However, despite being a relatively small fraction of the total organic matter found in most soils, the microbial biomass is potentially one of the main sources of P solubilized after such stresses.

Significant decreases in soil microbial biomass following drying have been reported, (Blackwell *et al.*, 2009a; Bottner 1985; Kieft *et al.*, 1987; van Gestel *et al.*, 1993), but evidence for the recovery of biomass following rewetting is varied. Grierson *et al.* (1998) reported increased biomass P in a spodosol 28 days after a drying–rewetting event, while Mondini *et al.* (2002) observed that biomass in a soil did not recover to original, predrying conditions 12 days

after rewetting, while Butterly *et al.* (2009) observed decreasing recovery following repeated drying–rewetting cycles. Various factors influence biomass recovery following perturbations and these are discussed below.

Reports on the effects of freezing–thawing cycles on soils vary greatly and are often contradictory. Large decreases in bacterial numbers following soil freezing are reported by Logsdail and Webber (1959) and Soulides and Allison (1961). They also reported that drying soils generally resulted in greater mortality of soil bacteria than freezing. In contrast, both Bolter *et al.* (2005) and Koponen *et al.* (2006) report little changes to the soil microbial biomass following freeze–thaw cycles. The contrasting findings are probably largely due to the different characteristics of the soils studied (Henry, 2007), the latter two examples being in soils from Northern Europe in which the majority of the biomass is likely to be adapted to surviving freezing conditions (i.e., it is psychrophilic, Robinson, 2001).

Generally, following both drying–rewetting and freezing–thawing it is thought that most of the nutrients released by microbial regulation and/or the microbial necromass are recycled by the surviving biomass in soils (DeLuca *et al.*, 1992; Dörsch *et al.*, 2004). Only recently has attention been given to the possibility that P originating from the biomass could be an important source of P in surface waters. Turner and Haygarth (2001) showed a relationship between the rewetting of air-dried soils and P solubilization from the biomass, and suggested a link to the pollution of watercourses. McNeill *et al.* (1998) and Turner *et al.* (2003b) also suggest that bacterial cell lysis represents a major source of mobilized P in Australian grassland soils, and a potentially important source of P available to plants. Turner *et al.* (2003a) postulated that the source of organic P found in streams in upland northern England originates from the biomass, being released in pulses following drying–rewetting and freeze–thaw cycles and is crucial to the maintenance of the trophic status of upland streams. However, conclusive evidence that there is a direct link between P release from the microbial biomass and P in surface waters is yet to emerge.

3.2. Solubilization, mobilization and transfer of phosphorus

Typically the solubilization of P in a soil and its mobilization and transfer from a soil to surface waters are considered as separate issues, but here we try to consider them as a continuum, while recognizing that they are distinct processes. It is important that this distinction is made clear. Solubilization refers to the release of previously bound or fixed P (such as that immobilized in the microbial biomass) into solution, while mobilization refers to the initiation of movement within the soil of this solubilized P as well as P-containing compounds and particles, including whole soil microbes. Both forms of P may be transferred from the soil to the aquatic environment, or may simply be cycled within the soil–plant system. The concept of

source–mobilization–transport to water is called the transfer continuum and discussed in further detail elsewhere (Granger *et al.*, 2010; Haygarth *et al.*, 2005a; Withers and Haygarth, 2007), where solubilization, detachment, and incidentals are all constituents of mobilization. This paper focuses mainly on the solubilization of microbial P. The mobilization and transfer of biomass P from the soil requires suitable energy, and the rewetting process in dried soils or thawing of frozen soils sometimes has the potential to deliver simultaneously conditions conducive to the solubilization, mobilization, and transfer of biomass P.

The movement of P from the biomass to surface waters is a three stage process. Firstly, the P must be released from the microbial biomass and made available for transport, a form of mobilization known as solubilization (Haygarth and Jarvis, 1999; Haygarth *et al.*, 2005a). Secondly, the solubilized P must actually move within the soil as a result of hydrological forces. Thirdly, the solubilized and mobilized P must be transported through the soil to the surface waters, connected via a range of transport pathways across a range of scales (Haygarth *et al.*, 2000; Haygarth *et al.*, 2005b). At first consideration, the actual processes of drying–rewetting and freezing–thawing in soils appears simple, and indeed, in reductionist experiments this is typically the case (Haygarth *et al.*, 2005a). Here, the only factors that need to be considered are the degree, duration, and rate of drying–rewetting and freezing–thawing, but many factors across a range of scales influence soil nutrient processes (Groffman, 1994) including whether mobilized biomass P will be transferred to surface waters. For example, at the global and regional scale, factors such as climate and land use will be influential, while at the landscape and field scale variations in geology, soil type and hydrology become important. The issue of delivery is discussed eloquently by Beven *et al.* (2005). However, factors of key importance at the organism and microsite scale, and which are considered here are:

(i) Soil structure
(ii) Soil chemistry (i.e., adsorption and/or desorption potential)
(iii) Soil type (particle size)
(iv) Vegetation status (quantity, stage of growth)
(v) Microbial spatial distribution
(vi) Small scale hydrological factors (rate of rewetting, quantity of water, soil water potential)

The first five factors are essentially intrinsic soil factors, while the latter is essentially extrinsic. Collectively these factors are referred to as hydropedology, the study of how water functions as both a catalyst for pedogenic processes within soil and as a vehicle for the transport, redistribution, and export of materials and energy between soils (Schoeneberger and Wysocki, 2005). This function is crucial in determining whether there is a direct contribution from the biomass to P in surface waters.

The rate of rewetting of a dried soil also influences how much P is actually leached from the soil. Blackwell *et al.* (2009a) found that, after rewetting a dried soil with equal quantities of water over periods of 0, 2, 4, 24, and 48 h, concentrations of dissolved P were highest in the 2-h treatment, and mostly in organic forms. While they could not directly link this P to a decrease in soil microbial biomass, it is likely that it was a major contributor. They attributed the differences in P concentrations in leachate caused by the different rates of rewetting largely to the opportunity for surviving microbes to recycle the solubilized P, which would increase with rewetting time. If the rate of rewetting is slow and there is a lag between soil rewetting and initiation of leachate flow, then there will be more opportunity for the surviving soil microbes and plants to utilize the available P, thereby reducing the quantity available to leave the soil in leachate. Others confirm that these P recycling processes can be very rapid, with Bushby and Marshall (1976) reporting the repair of bacterial cells damaged by desiccation occurring within an hour of rehydration, while De Nobili *et al.* (2006) report ''appreciable'' synthesis of ATP only 6 h after the rewetting of dried soils. Additionally, the rate of rewetting and leachate flow initiation will affect the ability of the soil to adsorb any released P, a factor which will also be affected by the soil chemistry and the chemistry of the P compounds mobilized. Similarly, the quantity of frozen water in a soil and the rate of thawing will affect whether leachate flow is initiated upon thawing, and how much time is available for solubilized biomass P to be taken up by surviving organisms or whether it leaves the soil via leachate. Opportunities for plants to take up solubilized P are also likely to increase with longer rewetting and thawing rates due to increased residence time of microbially derived P in the soil prior to leaching.

The relationships between P solubilizing bacteria and P uptake by plants are covered extensively elsewhere (Adesmoye and Kloepper, 2009; Goldstein, 1986; Tandon, 1987). Most of the work focuses on the ability of microbes to release mineral P by producing organic acids, which either reduce the pH of the soil and directly dissolve the mineral P, or cause chelation of iron and aluminum ions associated with P (Bajpai and Sundra Rao, 1971; Gyaneshwar *et al.*, 1998; Sperber, 1957). Either way, the mineral P becomes available to plants. However, few recognize the role of microbial uptake and subsequent release as a potential source of plant, although Gyaneshwar *et al.* (2002) recognizes that some non-P-solubilizing soil bacteria can be important as biofertilizers due to their ability to assimilate and subsequently release P upon death.

Phosphate is readily adsorbed in most soils so that it is often considered almost immobile (Sample *et al.*, 1980). However, many organic P compounds such as sugar phosphates and phosphate diesters have weak sorption capacity and are therefore more likely to leach (Frossard *et al.*, 1989; McKercher and Anderson, 1989), although higher order inositol

phosphates are strongly retained in soils (Turner, 2005). If the quantity of rewetting water or melt water is insufficient to stimulate leachate flow, then the form of P released by the biomass and whether rewetting or thawing occur rapidly is irrelevant in this context because any solubilized P will remain in the soil.

Rates of rewetting and thawing and the initiation of leachate flow are all affected by soil structure, with well structured soils being likely to rewet and thaw more heterogeneously than poorly structured soils, with the former also providing preferential pathways through which leachate flow is more likely to be generated (Grant *et al.*, 1996). For example, rewetting may affect only the surfaces of aggregates in the short term. This may cause a release of some biomass P which could flow down macropores and leave the soil in leachate (Preedy *et al.*, 2001). However, much of the soil may remain dry, only gradually wetting as water slowly permeates aggregates, thus there would be little opportunity for the P compounds to interact with the soil or enzymes. This slow permeability may mean that cell lysis is avoided due to slower rehydration of cells, but also may mean that instead of being flushed from the soil, any solubilized biomass P has a residence time long enough for it to be recycled by other soil organisms and remain in the soil. The same would largely be true for frozen soils undergoing defrosting, with the centers of aggregates potentially staying frozen for longer than the surfaces. Soil structure will also affect the spatial distribution of the microbial population, with populations less able to survive drying–rewetting and freezing–thawing probably being found in the center of aggregates, while those better able to survive perturbations may be dominant in more vulnerable zones (e.g., edges of aggregates). The spatial distribution of the biomass and the different components of the microbial community within the soil and within soil aggregates will also influence the effect of drying–rewetting and freezing–thawing cycles. For example, soil fungi tend to be located on the surfaces of soil aggregates (Hattori, 1988), and therefore are potentially more vulnerable to drying–rewetting and freezing–thawing episodes than bacterial communities which may be dominant in the center of aggregates. However, as discussed previously, fungi are more resistant to the effects of drying and rewetting than, for example, gram-negative bacteria, but more susceptible to freezing and thawing. Consequently the spatial distribution of microbial species within a soil will, at least partially, be controlled by soil hydrological and temperature distribution characteristics.

Soil structure is affected by many factors including particle size, which has also been shown to affect microbial vulnerability to desiccation, with microbes able to survive desiccation more readily in clayey soils than in sandy soils (Danso and Alexander, 1974; Marshall, 1964), although van Gestel *et al.* (1996) showed that this protection by clay is diminished under severe drying conditions (water potential less than −117 MPa). Erosion of soil as a consequence of rainfall events is another potential

mechanism for the transfer of biomass P to surface waters and does not require the solubilization of biomass P (Butler and Haygarth, 2007; Haygarth and Jarvis, 1999; Heathwaite *et al.*, 2005; Sharpley and Smith, 1990). Drying–rewetting and freezing–thawing has been shown in some cases to destabilize soil aggregates and so increase erosion (Denef *et al.*, 2001; Edwards and Cresser, 1992), but the relative importance of particulate transportation processes versus the dissolved fraction is not yet resolved.

The relative significance of the solubilization and transfer of nutrients to surface waters as a result of freezing and thawing and drying and rewetting processes will vary temporally and spatially. Undoubtedly, drying–rewetting will be more prevalent in summer, while freezing–thawing will occur more in winter. The fact that inherently, drying periods are likely to coincide with low river flows, means that the impact of solubilization and transfer of P from the biomass to surface waters at this time of year is likely to be more significant with regard to eutrophication than freezing–thawing events as less dilution is likely to occur. However the likelihood of transfer actually occurring could be minimal, due to the capacity of the soil to absorb large quantities of moisture when it is dry, and the increased likelihood of assimilation by plants and microbes which are active when it is warmer. On the other hand, freeze–thaw events are likely to coincide with periods of greater soil moisture content in the winter, meaning transfer via leachate flow is more likely. Also, at lower temperatures there would be less biological uptake of nutrients, meaning that potential for transfer is increased. However, increased discharge of rivers in winter could mean the general impact on surface waters with regard to eutrophication is likely to be less.

4. Uncertainties

Despite growing evidence for the role of the microbial biomass to act as an important source for P in soils and potentially to transfer to surface waters, there are still a number of factors which contribute towards the uncertainties associated with this topic. Here we address some of these.

4.1. Measurement of soil microbial biomass phosphorus

It is very difficult to separate the different effects of drying, freezing, osmotic regulation and osmotic shock upon P release from the biomass. This is largely due to the manner in which P soil is commonly measured, which generally involves extraction with an aqueous solution. Thus, determining whether the drying or rewetting process is the most significant is not possible, as rewetting is a fundamental part of the analytical method to

measure biomass P in air-dried soil. It cannot be measured directly under these conditions (Brookes *et al.*, 1982). While it is likely that desiccation and freezing are the main causes of cell mortality, they are not necessarily the causes of cell lysis and release of cytoplasmic material. Salema *et al.* (1982) reported that, after drying, rhizobial bacterial cells did not rupture but were morphologically similar to fully hydrated, live cells; only upon rewetting did cell rupture occur. Although it is difficult to determine which process is potentially most important in soils, it could be argued that this is an academic matter, and that drying will always inevitably be followed by rewetting at some point. The most important factor is the characteristics of the rewetting and potential for transport of the solubilized P either within or out of the soil. This is discussed further below and in Fig. 4.

4.2. Scales of measurement of impacts of abiotic perturbations

Most laboratory experiments eliminate many of the potential factors controlling the transferability of P from the system, for example, soil structure (spatial patterns of drying and microbial community distribution), and plant roots (uptake of P). Many of the experiments that have suggested that drying–rewetting and freezing–thawing of soils results in the transfer of biomass P to surface waters have used homogenized, finely sieved soil samples in which drying–rewetting and freezing–thawing is uniform.

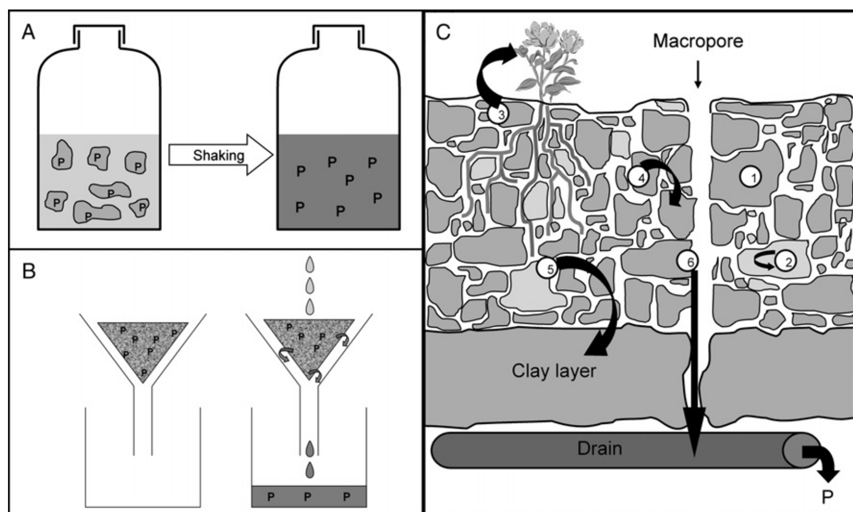


Figure 4 Factors affecting the measured quantity of P solubilized in drying–rewetting experiments.

While such experiments are important for determining the significance of certain processes, they do not reflect the natural system, and direct links between microbial P concentrations and increases in dissolved organic P in leachate following drying–rewetting and freezing–thawing cycles still needs to be demonstrated in the natural environment. Some of the factors involved are illustrated in Fig. 4A–C, which in combination show how laboratory experiments are likely to greatly overestimate the quantity of biomass P that may be solubilized, mobilized, and transferred from soils to surface waters.

Figure 4A illustrates that water extraction of dried soil will provide optimal results for the release of P from the biomass because any potentially solubilizable P will be extracted. However, the P in the extract does not necessarily represent P that would be transported in leachate. Figure 4B shows how leaching experiments using homogenized dried soil may also show high quantities of biomass P potentially being released and leached. The absence of large aggregates means the soil will be uniformly rewetted, and there are few opportunities for binding or uptake. However, the quantities of P released are unlikely to be as great as those in extractions, as some P will remain in the soil solution upon cessation of leaching. Another problem in assessing the amount of P solubilized in different drying–rewetting experiments is the characteristics of the drying and rewetting process (see above), and the P fraction which is measured. In Table 1 data are presented from several experiments investigating the effects of drying soils on solubilization of P. It can be seen that it is difficult to make easy comparisons amongst these data because of the different fractions, units, and methods used to establish the results.

Figure 4C illustrates some hypotheses that may explain why, upon rewetting a dried soil, biomass P may not be solubilized, but if it is, not necessarily transferred to surface waters. Six scenarios for individual soil microbes located in different parts of the soil as indicated in the diagram are considered:

1. A microbe in the center of a large soil aggregate is relatively unaffected by soil drying as the center of the aggregate remains moist. Some osmotic regulation may occur but no P is released.
2. A microbe in the center of a smaller soil aggregate is severely affected by drying and releases P as a result of desiccation and mortality, or cell lysis upon rewetting. However, there is no exchange of water from the center of the aggregate, so the solubilized P does not leave the system but is taken up by the surviving microbes.
3. A microbe on the edge of an aggregate releases P upon drying due to cell lysis. The P is mobilized and is taken up by plant roots which it contacts.
4. A microbe on the edge of an aggregate releases P upon drying by cell lysis. The P is mobilized and taken up by other soil microbes lower in the soil profile.

Table 1 Examples of measured effects of soil drying on P solubilization

Authors	Number of soils investigated	Phosphorus fraction measured	Concentration in moist soil	Concentration in dried soil	Change in concentration (%)	Contribution by microbial biomass (%)
Bartlett and James (1980)	1	Water extractable total P	ca. 5 ppm	ca. 32 ppm	ca. 540	
Qiu and McComb (1995)	1	Water extractable reactive P	8.7 μg P g^{-1}	11.0 μg P g^{-1}	37	98
Sparling *et al.* (1985)	15	Bicarbonate extractable reactive P	3.1–73.5 μg P g^{-1}	5.8–95.2 μg P g^{-1}	0–184	4–75
Turner and Haygarth (2001)	29	Water extractable total P	–	–	185–1900	–
Turner and Haygarth (2003)	29	Bicarbonate extractable total P	13.7–65.8 mg P kg^{-1}	22.2–88.1 mg P kg^{-1}	13–105	–
Turner *et al.* (2003b)	2	Water extractable total P	0.95–2.81 μg P g^{-1}	1.37–4.57 μg P g^{-1}	45–545	88–137
Styles and Coxon (2005)	37	Water extractable total P	1.15–33.74 mg PL^{-1}	–	−36–1010	–
Blackwell *et al.* (2009a)	1	Bicarbonate extractable reactive P	12.4 mg P kg^{-1}	21.3 mg P kg^{-1}	72	

5. A microbe on the edge of an aggregate releases P upon drying by cell lysis. The P is mobilized and avoids uptake. However, upon reaching a clay-rich layer of soil lower in the profile, the P is adsorbed.
6. A microbe on the edge of an aggregate situated adjacent to a macropore releases P upon drying due to cell lysis. Upon wetting the released P is washed down the macropore and avoids all interception mechanisms, enters a field drain and discharges into surface waters.

4.3. Spatial variability of microbial phosphorus

Generally, the biomass of P declines with depth (Blackwell *et al.*, 2009b; Fang and Moncrieff, 2005; Ross *et al.*, 1996). It is the surface of most soils that are likely to experience the greatest perturbation, so this is significant with regard to solubilization of P from the microbial biomass. However, there is little published data on the spatial variability of biomass P at the field scale. This is an important factor in understanding how different areas may solubilize and mobilize different quantities of biomass P, and ultimately where it is located within the landscape could affect its ability to transfer from the soil to surface waters. The lack of spatial data is probably due in part to the fact that the conventional method for measuring biomass P using the fumigation-extraction method (Brookes *et al.*, 1982) can be time consuming, particularly when processing large numbers of samples. Consequently, when measuring soil microbial biomass P at a field scale, typically several soil samples are collected to account for spatial heterogeneity and these are subsequently bulked prior to analysis in order to reduce sample preparation and analysis time (e.g., De Nobili *et al.*, 2006; He *et al.*, 1997; Perrott *et al.*, 1992). While this may provide an accurate mean estimate of properties at the field scale, information on the geo-variability of biomass P across a field is lacking.

As part of a project investigating temporal variability of soil microbial biomass, we compared a relatively rapid bulked sample method with a more rigorous, time consuming Latin square sampling +1 design (LSS+1) (Munholland and Borkowski, 1996) for measurement of mean ($n = 26$) soil microbial biomass phosphorus (biomass P) in a grazed pasture in Devon, England. Mean biomass P values of 175 and 218 mg kg^{-1} were measured using the LSS+1 method and the bulked sample method respectively, showed a difference of some 20%. The distribution of the biomass P in the field using the LSS+1 method is shown in Fig. 5, with the "hotspot" towards the south of the field explaining much of this difference. This hotspot was attributed to a patch of the field which was particularly wet at the time of sampling, and the implications for the drying and subsequent rewetting of this area as a potential source of microbial P are much greater than for other parts of the field. However, its significance with regard to

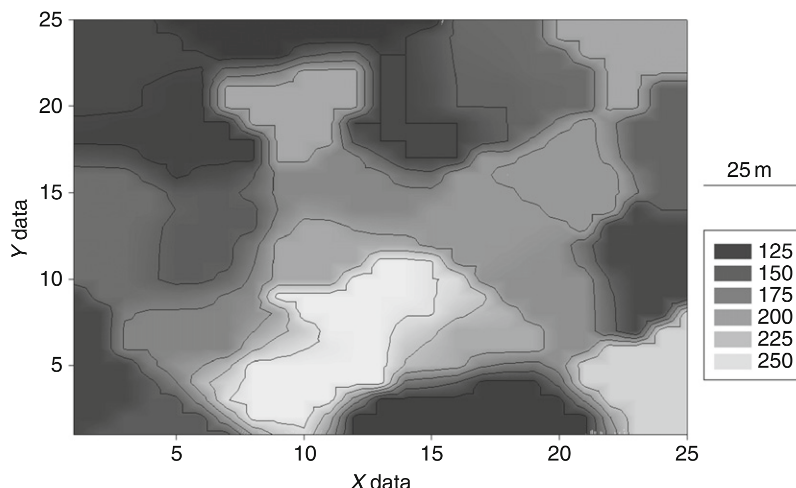


Figure 5 Distribution of soil microbial biomass P measured in a 1-ha long-term pasture, mg P kg^{-1}. *Y* data represents northings, *X* data eastings.

potential for transfer to surface waters will depend upon its hydrological connectivity, and potential pathways as described above.

4.4. Relative contributions of sources of phosphorus in soil

While some attempts have been made to quantify the sources of P in soil solubilized (e.g., Sparling *et al.*, 1985; Turner *et al.*, 2003b) and transferred from soils following abiotic stress (Blackwell *et al.*, 2009a), further work is needed to establish the significance of the release and loss of P from the soil microbial biomass more widely. This may be possible by investigation of specific forms of P in leachate following abiotic stress. For example, Turner *et al.* (2003b) speculate that if recently lysed cells are contributing significant amounts of P in leachate then this could be reflected by the quantities of partially degraded nucleic acids or phospholipids (labile orthophosphate diesters) present.

While several researchers report the microbial biomass making a large contribution to solubilized P following abiotic perturbations (Sparling *et al.*, 1985; Turner *et al.*, 2003b; Qiu and McComb, 1995), the percentage contribution varies greatly in different soils (Table 1). Some researchers have shown that in some soils the microbial biomass actually contributes very little to the solubilized P following abiotic stress and that the primary source of the solubilized P is in fact from nonliving organic matter in soils (Butterly *et al.*, 2009; Magid and Nielsen, 1992). However, in soils under pasture with $>2\%$ organic matter and relatively low bicarbonate extractable P values, generally

there is likely to be a significant contribution to solubilized P following abiotic stress from the microbial biomass (Sparling *et al.*, 1985).

Relative contributions of different fractions of the soil microbial biomass will vary. For example, in some soils fungi may be more prevalent, while in others bacteria may have a greater significance. The relative role of the different components of the soil microbial biomass would be useful to know. Additionally, microbial biomass C:P ratios are much more variable (range ca. 12–36) (Brookes *et al.* 1984) than biomass C:N ratios (Jenkinson, 1988). These apparent variations in the C:P ratios of the biomass under different managements could be an important regulator of the P released from the biomass through different biotic and/or abiotic processes. It would seem likely that biomasses with smaller C:P ratios would release more P per unit of biomass C than those with higher ones. However, this awaits investigation.

5. The Significance of Changing Climate and Soil Management

Understanding the processes and mechanisms controlling the solubilization of biomass P within the soil and transfer to surface waters will provide a better understanding of how both soil nutrient dynamics and surface water quality may be affected by changing climatic conditions. In addition, the management of solubilization of biomass P by drying–rewetting in agricultural soils could be a useful tool in minimizing fertilizer use and maximizing yield (Chepkwony *et al.*, 2001).

Prolonged soil dryness is increasingly likely to occur in some temperate regions as a result of global climate change (Intergovernmental Panel on Climate Change, 2001), which could affect organisms already adapted to dry conditions if their survival thresholds are exceeded. Specifically, the UKCIPO2 report (Hulme *et al.*, 2002) predicts that, in the UK, summers and autumns will become warmer and drier, with average temperature increases of up to 3.5 °C and summers up to 50% drier than now, predictions further confirmed in the Murphy *et al.* (2009) report. Although these changes are likely to be gradual, potentially providing the biomass time to adapt, the likelihood of longer drier periods in the summer could mean more extreme drying and rewetting events, potentially triggering the solubilization of biomass P in soils. Turner and Haygarth (2001) showed that P solubilization occurs more rapidly in soils dried at higher temperatures (30 °C compared to 20 °C). This has implications for the release of P into the surface of soils that commonly exceed 30 °C in temperate regions (Russell, 1988), meaning that a change in climate to shorter periods of warm dry weather interspersed with heavy rainfall could result in increased

mobilization of microbial P and transfer to surface waters. Similarly, patterns of freezing–thawing of soils may alter, particularly in higher latitudes, where the effects of freezing–thawing cycles on the microbial biomass and nutrient turnover are finely tuned to the seasonal temperature patterns (Mikan *et al.*, 2002). With regard to soil management, sometimes the priority will be to reduce P loss from soils to surface waters. At other times the key issues will be either to maintain soil P concentrations at sufficient levels to enable sustainable plant/crop growth, or maximize biomass P solubilization at key times in order to provide a P supply to plants.

A better understanding and consideration of the solubilization mechanisms of biomass P could help devise more efficient irrigation patterns and land management techniques (e.g., vegetation cover, soil structure management) with regard to the cycling and leaching of P. Key factors with regard to the latter are associated with the timing of biomass P solubilization events and the ability of crops to access this large and important pool of P in the soil (Chepkwony *et al.*, 2001). Management techniques that can moderate drying of soils may then be considered in this respect. For example, tillage that reduces the drying of soils could prevent large releases of biomass P upon rewetting, so/thereby reducing potential transfer to surface waters, while in other situations it may prove beneficial to reduce soil moisture levels prior to rewetting in order to stimulate release of microbial P to make it available to plants. Also, the timing of applications of slurry, which can also have a rewetting effect, should be considered carefully. However, such management will not be easy, and unless made in controlled environments (e.g., irrigated land) will be highly dependent on reliable long-range weather forecasting. Additionally, before such approaches can be adopted, a better understanding of the relationships, quantities of P and timeframes involved in these processes is required. Some evidence suggests that immobilization by the biomass of solubilized P in soils that is not leached is rapid and therefore only very short windows of opportunity for plant uptake may exist. However, if found to be significant, it could mean that a fundamental change occurs in the way agricultural practices regulate soil moisture, with soil moisture management playing a crucial role in nutrient availability and crop productivity.

Turner *et al.* (2003a) indicated a potential link between the biomass P in upland soils and P concentrations (and hence ecology) of upland streams in the UK. These regions are thought to be some of the most vulnerable ecosystems to climate change, and disruption of the potential linkage between the biomass P and P concentrations in these surface waters could contribute to the overall effect of changing climate on these ecosystems. Qiu *et al.* (2004) also link soil biomass and soil drying–rewetting events with elevated concentrations of P in lakes in south-western Australia. Laboratory experiments have alluded to this link (Turner and Haygarth, 2000, 2001), but still no direct evidence exists.

6. Conclusions and Future Work

There is substantial evidence that abiotic stress of soils through drying–rewetting and freezing–thawing contributes to the solubilization of soil nutrients, including P, some of which is likely to originate from the soil microbial biomass. Soils with different characteristics and microbial communities will react differently to changing patterns of drying–wetting and freezing–thawing. It is therefore important that we understand how diverse soil types (e.g., boreal and temperate soils) are likely to react with regard to biomass P cycling and potential P loss to surface waters under these conditions. There are still many unanswered questions with regard to the direct connectivity between biomass P and surface water P concentrations. Solubilization of biomass P is complex depending on many intrinsic and extrinsic soil factors including the manner of the rewetting and freezing processes, the size, location, activity, and stage of growth of the soil microbial biomass community, and whether or not the population is adapted to drying–rewetting and/or freezing–thawing cycles. There are many contradictory reports on the effects of these perturbations on the microbial biomass and the release of biomass P, but generally it is likely that only extreme, unusual conditions will illicit large responses with regard to its solubilization and transfer, as otherwise much P would be regularly lost from these systems and soil microbial populations would decline. The transfer of biomass P to surface waters depends upon the complex spatial and temporal interactions among hydropedological factors (drying, rewetting, freezing, thawing and flow paths) and microbial biomass populations and communities. Consequently it is hypothesized that only a small proportion of P solubilized by drying–wetting and freezing–thawing events is likely to find its way from the soil to surface waters, but this is not to say that this small proportion is not significant with regard to surface water quality and nutrient loss from the soil.

Much of the work into the effects of rewetting dried soils and freezing and thawing has used prepared soil which has been uniformly subjected to extreme treatments. In reality it is likely that for many soils, only the surface of the soil will be affected to any extreme. The actual impacts of surface effects as opposed to whole-soil effects require further investigation, along with the impacts on intact soils and soils containing flora and fauna. Identifying the sources of the P mobilized in leachate is another challenge, along with determination of the relative roles of native soil organic matter and the soil microbial biomass. New approaches are required for these investigations, and these include measurement of the forms of P in leachate (such as nucleic acids and phospholipids) and isotope tracing of P compounds, although these methods have their limitations.

Changes in patterns and degrees of abiotic stresses as predicted by climate change scenarios could have significant effects on the cycling of P (and other nutrients) in the soil, and in particular in losses of P from the soil to surface waters. As we have described here, some components of the soil microbial biomass are resilient to abiotic stresses, while others are particularly vulnerable. We have described methods by which the contribution of the microbial biomass to solubilized and mobilized P in leachate from dried soils could be investigated, but considerable work is required on the characterization of the impacts of different patterns of drying–rewetting and freezing–thawing on the biomass in different soils before these estimates can be accurately quantified. If conditions prevail which facilitate the transfer of P solubilized from the soil microbial biomass to surface waters, then not only could there be serious implications for the quality of surface waters, but also there could also be related issues concerning the loss of P from soils. As concerns grow over global food security and nutrient optimization, it is important that we better understand these processes and how they will be impacted by climate change.

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CHAPTER TWO

Improving Productivity of Crops in Water-Limited Environments

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Contents

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Abstract

This review deals with improving the performance of dryland crops in dry, mainly semiarid, environments. Although such crops are often limited by water, the development of the notion of water-limited potential yield has shown that their yields are often limited strongly by other factors. These factors are explored by dissecting the water-limited potential yield into a framework involving four largely independent components, namely, the potential water supply for the crop; the fraction of that water supply that is transpired; the efficiency with which the crop exchanges water for CO_2 in producing biomass (Transpiration Efficiency); and the fraction of the biomass that ends up in the grain (Harvest Index). This framework is used to explore a wide range of agronomic possibilities for managing crops so as to get close to the water-limited potential, including managing previous crops, forages and fallows to increase soil water at sowing; reducing evaporative losses from the soil surface; ensuring that sowing and flowering occur at the right times; maximizing soil water extraction by the crop; and ensuring that there is adequate water available during late floral development and grain filling. Such operations often involve trade-offs and risks that must be managed.

1. Introduction

At the beginning of the twenty-first century, the world contained 1.5×10^9 hectares of cropland. Of this, 74% was rainfed and produced 60–65% of the world's food supply (Biradar *et al.*, 2009; Döll and Siebert, 2002). During the previous 50 years, the area of irrigation increased remarkably, particularly in developing countries, and is now responsible for 87% of the world's consumptive use of water (Shiklomanov, 1999). Yields of crops growing on irrigated and favored rainfed land now approach 80% of the yield on experiment stations. The 2008 World Development Report concludes that there is little scope to close the yield gap further on this land (World Bank, 2008).

Future water allocations for irrigation will come under increasing competition from cities, manufacturing industries and the needs of the environment. Rainfed crops mostly produce grains for human food and stock feed whereas irrigation produces high-value crops such as vegetables, as well as grains. Dwindling irrigation supplies are likely to be allocated to high-value crops rather than grains. The productivity improvement of rainfed grain crops, particularly those grown in the less favorable rainfed environments will have to increase to meet the increasing world food requirement.

Rainfed cropped lands vary from semiarid regions where the water supply fails to meet the evaporative demand in most seasons, to humid regions where the water supply is adequate in most seasons. Here we concentrate on crops grown in semiarid and subhumid regions (Thornthwaite, 1948).

Much of the review concentrates on wheat, and many of the examples are from Australian environments. The conclusions for wheat generally apply to other cereal species except perhaps in relation to soil-borne diseases (see Section 4.3). The semiarid and subhumid climates of Australia differ from those on other continents, particularly during the winter, which is milder than in continental climatic regions; during the growing season of wheat, however, there are many similarities. The environments relevant to this review are the areas in Africa and Asia covered by the international centers ICRISAT and ICARDA, the Great Plains of north America, the mixed farming areas of Australia, western China, the steppe of south-west Europe and west Asia, the drier parts of the pampas of Argentina and the Mediterranean basin of Europe. In some of the subhumid parts of these regions, crops are increasingly being grown continuously, but in semiarid environments many crops grow as phases of a rotation with fallow, pastures, or forages. In this review, we concentrate on crops harvested for grain and refer to forages only in so far as they affect the grain crops with which they grow in phased rotation.

Crop in these regions have lower and more variable yields than those in irrigated and favored rainfed regions, but they do have some innate advantages. Their production comes without the large public costs of irrigation structures and depletion of groundwater. Grain produced under terminal drought is usually of high quality and does not require drying. In this review, we discuss the prospects of increasing productivity.

2. Water-Limited Potential Yield

While it seems obvious that crop yields are typically limited by water in semiarid environments, the development of the notion of water-limited potential yield has shown that yields are often limited strongly by other factors. Analyses of wheat yields by Angus and van Herwaarden (2001), Cornish and Murray (1989), French and Schultz (1984), Grassini *et al.* (2009), and Sadras and Angus (2006), foreshadowed by (Trumble, 1937), have shown that farmers' yields generally fall well below what one could expect if water were indeed the main limitation. Figure 1, whose use was pioneered by French and Schultz (1984), schematically illustrates this experience. The sloping line depicts the expected yield if water, well distributed through the growing season, were the only limitation. This line has a slope of about 20 kg ha^{-1} mm^{-1} and represents the maximum yield per mm of water transpired by the crop, that is, the water-limited potential yield. This slope is slowly being increased by classical breeding and presumably also by rising levels of atmospheric CO_2. The cloud of points depicts the range of experience. The intercept on the *x*-axis is usually interpreted to be the (least) loss of water by direct evaporation from the soil (Hanks *et al.*, 1969; Leggett, 1959), a point that we will expand on later (Section 4.2.1).

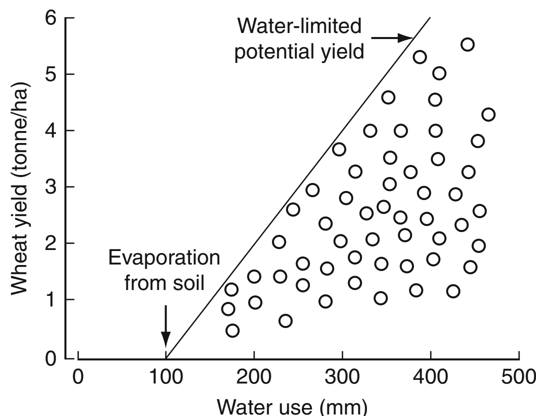


Figure 1 Schematic portrayal of wheat yield in relation to seasonal water supply (rainfall during the growing season plus available water in the soil at the time of sowing). The solid line depicts yield if water is the only limitation. It has a slope of about 20 kg ha^{-1} mm^{-1}. The intercept of that line on the *x*-axis reflects the loss of water by direct evaporation from the soil. The points cover the range of farmers' experience and are typically well below the solid line because of yield limitations due to weeds, disease, poor nutrition, frost, and other problems. Adapted from Passioura (2002) with permission of CSIRO Publishing.

Poor distribution of rain can account for some of the yields falling well below the line, but there are many other reasons as well: untimeliness of operations (especially late sowing), weeds, poor establishment of seedlings, disease, inadequate nutrition, frost, heat, excessively acid topsoil, and subsoil that is inhospitable to roots because of sodicity, boron toxicity, salinity, acidity, or alkalinity. Some of these, especially root diseases and inhospitable subsoil, reduce the roots' ability to take up water and can be partly alleviated by more rain. In this sense, the crops are still water-limited, even though they are performing well below their water-limited potential.

The notion of water-limited potential yield has been an inspiration to farmers. In Australia, many farmers have now adopted it as a benchmark for crop yield. If they find that their crops are performing well below the benchmark, they look for the reasons and try to improve their management accordingly (Passioura, 2002).

This is not to say that achieving the water-limited potential is always an economically realistic aim (Hochman *et al.*, 2009b).The level of management and inputs required to do so may exceed an economic optimum, which will vary from region to region along with grain prices and input costs. In addition, aiming for too high a yield may increase the risk of a crop doing poorly if it runs out of water during the critical periods of flowering and grain filling. The following case study illustrates this point.

2.1. Risk management: A case study

Figure 2, adapted from Angus and van Herwaarden (2001), shows mean wheat yields in the shire of Wagga Wagga, a relatively high-yielding region of south-eastern Australia, in relation to growing-season rainfall, over the years 1949–1983—see also Cornish and Murray (1989). The solid diagonal line depicts a water-use efficiency (WUE) of 20 kg ha^{-1} mm^{-1}.

The simulated points in Fig. 2 come from the SIMTAG model of Stapper and Harris (1989). The simulation assumed that the crops were well-managed and were without disease. It thus illustrates the effect of poor seasonal distribution of rain on yield—for example, a large proportion of small falls of rain can lead to larger losses by direct evaporation from the soil (Sadras, 2003) or water deficits at flowering can lead to infertility even though the crop may have had a good water supply at other times. The simulation also illustrates the transition, at about 500 mm, to radiation rather than water being limiting when the rainfall is large.

The striking feature of Fig. 2 is that, except in the driest years, most of the field data fell well below not only the bounding line but also the cloud of simulated points. The horizontal distribution of these points suggests that water was not the main limiter of yield in most years, an observation similar

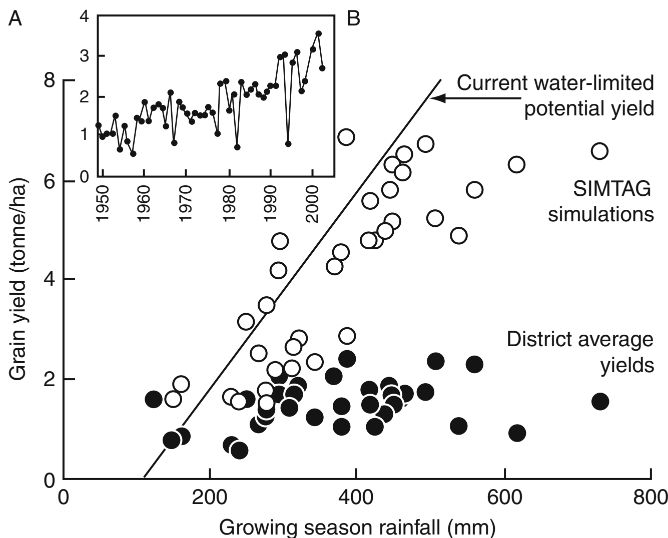


Figure 2 (A) Reported (•) and simulated (○) mean wheat yield in the shire of Wagga Wagga for 1949–1983 in relation to growing-season rainfall. The solid line depicts the upper bound of reported yields across a range of studies in southern Australia. It has a slope of about 20 kg ha^{-1} mm^{-1} (2.0 kg m^{-3}). Adapted from Angus and van Herwaarden (2001) with permission. (B) Reported wheat yield for Wagga Wagga shire, showing markedly increased yield since the 1980s.

to that of Rockström and Falkenmark (2000) for Sub-Saharan Africa. In fact, the main limiter seems to have been risk management. The collective experience of the farmers was that crops that grew too fast during their vegetative phase tended to produce low yields. This happened irrespective of whether they boosted crop growth with high rates of fertilizer or high plant density. Each intervention could lead to "haying-off," in which crops produce a large biomass, low yield and pinched grain (see Section 6.3). After experiencing hayed-off crops, farmers became cautious about applying high rates of fertilizer or seed to produce high yields. Their observations about nitrogen status were confirmed by numerous field trials in the region during the 1960s and 1970s that rarely showed profitable response to nitrogen fertilizer.

Subsequently, it was realized that cryptic root diseases were rife during this period. These exposed the crops to unexpectedly large water deficits during grain filling because of the inability of the roots to effectively exploit water in the subsoil (Angus *et al.*, 1991; Kirkegaard *et al.*, 1994b). Farmers learned how to control root diseases, mostly by growing break crops and removing grasses during the pasture phase because they were hosts to pathogens. They then had the confidence to aim for and achieve high yields by increasing their inputs, especially nitrogen. Many farmers are now achieving yields close to the upper bound (Angus, 2001).



3. Water as a Limiting Resource: An Analytical and Diagnostic Framework

Figure 1 illustrates both the water-limited potential yield and the losses of water by processes other than transpiration (mostly direct evaporation from the soil). It is not shown in the figure, however, that the slope of the bounding line can be dissected into the product of the harvest index (HI) (the proportion of the above-ground biomass that is in the grain) and the transpiration efficiency (TE) for total above-ground biomass (the above-ground biomass at harvest divided by total transpiration).

Thus, the water-limited potential yield can be dissected into several components to help answer the question of how one can manage a crop or improve its genetic makeup to increase the water-limited yield (Passioura, 1977). The crop can be managed, and/or bred, to:

- Increase the potential water supply
- Transpire more of that water supply
- Exchange transpired water for CO_2 more effectively in producing biomass
- Convert more of the biomass into grain

The product of the latter three constitutes an identity:

$$Y = T \times \mathrm{TE} \times \mathrm{HI}$$

where Y is yield, T is water transpired, TE is transpiration efficiency for producing biomass (i.e., biomass divided by water transpired), and HI is harvest index. Although these three components often interact, they are sufficiently independent to make it worthwhile considering them one by one (Araus *et al.*, 2002; Passioura, 2006; Richards *et al.*, 2002).

Historical increases in yields of new cultivars are testimony to previous genetic improvements in water-limited potential yield, and are especially associated with the introduction of semidwarf varieties with their substantially larger HIs. Some breeding programs have made use of the identity to target specific traits directed at raising one or more of the components (Richards *et al.*, 2002).

The greater immediate opportunity for improving water-limited yield, though, is in reducing the often large gaps between actual and potential yields, rather than in improving the potential. Here the identity is useful as a diagnostic tool, in that it can be used as a decision tree for analyzing the causes of, and finding ways of reducing, large gaps. For example, a large gap can result from low biomass ($T \times$ TE) or low HI, and the processes leading to these alternatives are usually markedly different. Of the three components in the identity, TE is the most robust. It varies by about 10% across cultivars in a given environment, and is little affected by nitrogen nutrition. HI can vary from 0% to 50%, but has a typical maximum of about 40% for wheat in dry environments. The loss component of the water supply can be at least double that shown in Fig. 1 if vegetative development is hindered. On the other side of the ledger, the amount of water in the soil at sowing can be augmented by the management of preceding fallow periods (Section 4.1). Water stored during these fallows is typically contained below about 60 cm in the soil profile, and can be especially valuable because it tends to be accessed during flowering and grain filling, which leads to large HIs. See Section 6.3.

In this review, we use the identity, expanded as illustrated in Fig. 3, to cater for the main components in the crop's water balance, as (1) an analytical framework for systematically discussing the factors influencing water-limited potential yield, and (2) as a diagnostic guide for identifying the reasons for large gaps between actual and potential yields. It is also useful diagnostically to distinguish between too few kernels per m^2 (KNO) and low kernel weight (KW) as factors contributing to a low yield. This alternative simple analysis of yield components is also shown in Fig. 3.

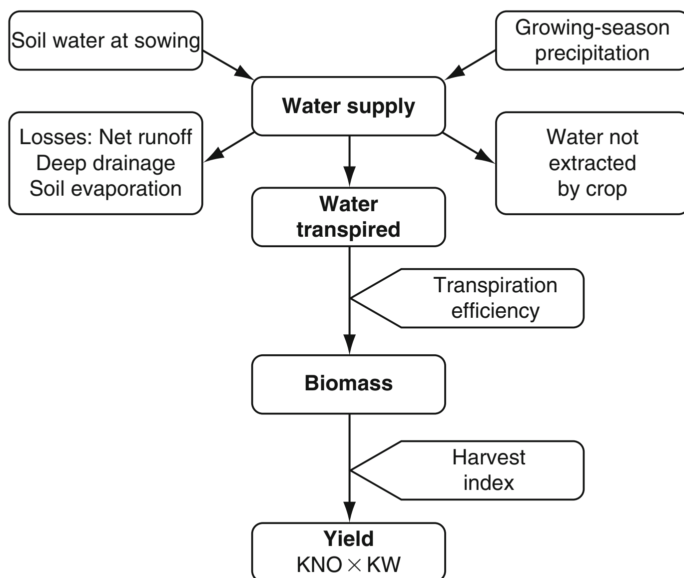


Figure 3 Schematic illustration of the main variables and processes involved in generating grain yield from a limiting water supply. Also included is an alternative dissection of yield into number of grains per hectare (KNO) and mean kernel weight (KW).

4. Water Balance and Transpiration

4.1. Available soil water at sowing

An important source of water for dryland crops is the residual water accumulated during a previous fallow or unused by the previous crop or forage. This amount can be as much as the water-holding capacity of the soil after an effective fallow or can be insignificant after depletion by a previous crop or if the fallow was weedy. In extreme circumstances, for example, a crop following a perennial forage crop, the amount can actually be less than the recognized lower limit for annual crops, and thus enters the balance sheet as a deficit that must be satisfied before the water available to the crop in a given soil layer can be accumulated.

Fallowing is normal practice in semiarid regions and some subhumid regions when the expected rainfall during crop growth is less than the amount required for acceptable yield. The meaning of fallow in this context is a field where plant growth is prevented or greatly reduced for a period as short as a few weeks to more than 1 year. Soil may be fallowed, by cultivation and/or the use of broad-spectrum herbicides. For fallows longer

than a few months, it is usually necessary to cultivate or spray herbicides more than once (Sims, 1977).

Soil water accumulated by a fallow is usually much less than the rain that falls on it. The efficiency of fallow storage is the increase in soil water expressed as a proportion of rain during the fallow period. Fallow efficiency is typically 25–30%, and varies with environment and management (Fischer, 1987; Peterson and Westfall, 2004). Reasons for low fallow efficiency are evaporation from the soil surface, runoff, and deep drainage. For dry environments runoff and deep drainage are usually much smaller than evaporation, though intense summer storms can give much runoff.

Evaporation from the soil is a two-stage process during which the water evaporates from a wet soil at a rate determined by net radiation and later, as the soil surface dries, by the ability of the soil to conduct water to the soil surface (Suleiman and Ritchie, 2003). During this second stage the amount of water evaporated from a bare soil proceeds as the square root of time. Thus it takes place ever more slowly, and in practice there is little evaporation from deeper than 0.3–0.6 m during a dry summer. For effective storage of soil water during a fallow it is important that water accumulates deeper than this range and that the rate of evaporation is minimized. The most effective method of reducing the rate of soil evaporation, at least for a few weeks, is to retain stubble from a previous crop (O'Leary and Connor, 1997; Verburg *et al.*, 2004). This extends the length of the first stage of drying, but eventually the surface will dry unless there is more rain within about 3 weeks, and evaporation will enter the second stage. Not all evaporation from the stubble is directly from the soil surface; part of the loss appears to be through the xylem of dead roots and stems (Ward *et al.*, 2009). Flattening the stubble with harrows breaks the continuity of this water path with a small water saving, provided the top parts of the stubble are not detached and blown away.

Stubble also impedes runoff and reduces sealing of the soil surface by the impact of rain drops, thereby increasing infiltration during rainfall (Foley and Silburn, 2002). A heavy stubble may also intercept a few mm of rain, thereby preventing it reaching the soil, though little information is available on this. Retained stubble is particularly effective in preventing movement of snow from a field so that fallow efficiencies in regions with cold winter are generally high (Campbell *et al.*, 1992).

Maintaining a fallow free of weeds is critically important for retaining water in the subsoil below the depth to which the soil is likely to dry by evaporation. Such water is typically accessed late in the season, when it can be exceptionally valuable in contributing to the yield. Every mm of it can contribute as much as 60 kg ha^{-1} to the yield (Section 6.3), so even a few deep-rooted weeds can be harmful.

In semiarid environments fallowing is usually a strategic practice, that is, fallows and crops alternate regularly, irrespective of seasonal and crop

conditions. An exception is the proposal of Fischer and Armstrong (1990) to start a fallow only if there is sufficient soil water to make a significant contribution to the following crop. Another exception to regular fallows is the system of opportunity cropping which originated in semiarid regions in eastern Australia where it is possible to grow both summer and winter crops (see Section 7.2).

Another successful technique for making good use of opening rains is to sow dry (Harris, 2006). This technique provides the best opportunity for early sowing and has been made possible because conservation tillage practices lead to sufficiently loose topsoil at the end of a fallow. While it seems risky, with the danger of a small fall of rain starting germination which then aborts, in practice this does not seem to be a problem in the regions where it is practiced. Partial germination following a small fall of rain may even be an advantage, in that it can activate enzymes that accelerate germination after a later fall of rain (Bradford and Bewley, 2002). The release of glyphosate-tolerant dryland crops further facilitates dry sowing because of the options it provides for postemergent weed control.

In some circumstances rain on the fallow may exceed the field capacity of the soil to the depth of a crop's rooting zone. This can lead to deep drainage, as indicated by leaching of nitrate and salt in eastern Australia (Turpin *et al.*, 1998) and nitrate in the Great Plains of the USA (Lamb *et al.*, 1985). In both these cases it appears that water accumulated below the root depth over several fallow cycles. Similar deep leaching of nitrate in western Australia is associated with summer storms on sandy soils (Asseng *et al.*, 2001b). In some landscapes, deep drainage can lead to outbreaks of dryland salinity in low-lying places (e.g., Ridley *et al.* (2001)), and thus careful management is needed to balance the requirements of both productivity and environment.

Where a fallowed soil profile is likely to fill with water or there is risk of erosion, it is possible to minimize excess water accumulation by growing a short-season cover crop such as millet during the normal fallow season and terminate its growth with a herbicide if the season then turns dry (Price *et al.*, 2006). In this case, the reduction in soil evaporation due to the soil cover more than compensated for water use by the millet and increased yield of a following wheat crop.

The supply of soil water is also affected by the previous crop or forage. In a hydrologically stable dryland farming system it may be imagined that each crop depletes the soil water to a reasonably uniform lower limit (Ritchie, 1981). In some circumstances dryland crops leave a surprisingly large amount of soil water at maturity (see Section 4.3), which may be accessed by later crops.

Residual water after forages is more variable than after crops. Annual pastures can be even less effective in extracting soil water than annual crops because of their shallow root systems and earlier maturity. In contrast, perennial pastures, especially those that contain lucerne (alfalfa), normally

extract more soil water than healthy annual crops, and extract that water from deeper in the soil than annual crops (Angus *et al.*, 2001; McCallum *et al.*, 2001). This dewatering ability of lucerne results from a long period of extraction rather than a particularly high extraction rate. The soil can be so dry at the end of a phase of perennial pasture that a large amount of recharge is needed before the water content returns to the generally accepted lower limit for annual crops. This amount can be an effective buffer for reducing deep drainage in wet environments (Halvorson and Reule, 1980; Ward *et al.*, 2002). It is also an indication of the risk of low yield in a dry season. In soils with high water-holding capacity the buffer is typically greater than 150 mm (Angus *et al.*, 2001) while in soil with relatively low water-holding capacity the buffer is less than 100 mm (Ward, 2006).

Partly offsetting the dewatering ability of lucerne is greater infiltration into the subsoil through taproot channels, which remain in the soil for several years after roots decay. McCallum *et al.* (2004) found that the average infiltration into subsoil after lucerne was 10 mm more than into subsoil where annual pastures or crops had grown. This amount is not enough to compensate for the buffer at the end of a lucerne phase but after the soil profile is recharged during the cropping phase, the greater infiltration provides a continuing benefit. In the experiments of McCallum *et al.* (2004) the value of lucerne biopores for infiltration was apparently because their diameter (4 mm) was much greater than for annual crops and pastures. Cresswell and Kirkegaard (1995) had earlier found no greater infiltration in soil after a canola crop than after a wheat crop because the diameter of root channels were not sufficiently different.

4.2. Evaporation from the soil and other nonproductive losses of water during the growing season

4.2.1. Influence of canopy development

In regions subject to intense rainfalls, run-off from the soil surface can be substantial and can discharge into drainage lines. In general, though, episodes of ponding that induce run-off lead to run-on in lower parts of a field (Batchelor *et al.*, 2002; Eastham *et al.*, 2000), so that there may be little net loss from the field as a whole, except perhaps through greater deep drainage from low-lying parts. Such lateral distributions, even when there is no net loss, do lead to spatial variability in water supply, an issue covered in Section 7.3.

Direct evaporation of water from the soil surface (E_s) is typically the main loss during the growing season. It is essentially an unproductive loss of water that can be reduced by management and vigorous genotypes, but not eliminated altogether. This loss is especially large in winter-rainfall regions, where it can exceed half of the seasonal water supply, which comes mostly from rain during the autumn to spring growing season (Cooper *et al.*, 1983;

Leuning *et al.*, 1994). E_s is fast when evaporative demand is high and when the soil surface is uncovered and moist, all common conditions during early growth. As a crop grows and the foliage covers the soil, more of the water is transpired and less evaporates. The roots dry the topsoil quickly and thereby induce an early transition to something similar to the second stage of drying (see Section 4.1). Denmead *et al.* (1997) have shown that the proportion of total canopy evaporation attributable to E_s falls exponentially with the green leaf area index, L, according to $\exp(-0.61\ L)$.

The intercept of about 100 mm shown in Fig. 1 is usually interpreted as an estimate of the evaporative loss, but it is better thought of as the least evaporative loss. The actual loss varies with soil type and stubble management (Fischer, 1987; Hatfield *et al.*, 2001), and with the seasonal weather. It tends to be greater where there are many small falls of rain (Sadras, 2003), and is smaller in very sandy soils (Sadras and Roget, 2004) which quickly enter the second stage of drying after a fall of rain. When a crop is grown predominantly on stored soil water, with little in-season rain, the estimated E_s is as little as 25 mm, representing little more than the subsurface water exposed by the sowing operation (Angus *et al.*, 1981).

E_s also depends strongly and inversely on the development of the leaf canopy (Fischer, 1979; Shepherd *et al.*, 1987). This is illustrated schematically in Fig. 4, which differs from Fig. 1 in showing biomass production

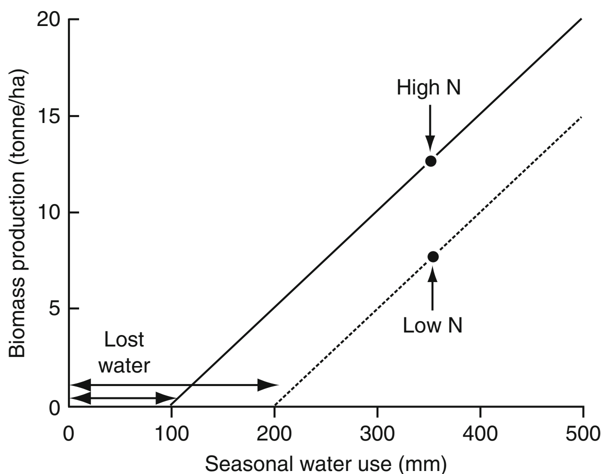


Figure 4 Biomass production in relation to a given seasonal water supply. The solid line represents a well-managed winter cereal crop. The dashed line represents a nitrogen-deficient crop. The slopes of these lines, TE for above-ground biomass, are 50 kg ha^{-1} mm^{-1} of water transpired, a reasonably robust value for winter cereal crops in Australia. The intercepts on the *x*-axis denote the water lost to the crops by evaporation from the soil surface (E_s). Adapted from Dunin and Passioura (2006) with permission of CSIRO Publishing.

rather than grain yield in relation to water supply. Here, two crops differing in N supply though having the same water supply vary greatly in canopy development and thence evaporative water loss. The slopes of the lines are similar, for transpiration efficiency (TE in Fig. 3) is essentially independent of N status (Condon *et al.*, 1992). Thus, in relation to Fig. 1, of the two crops represented by the points in Fig. 3, one would appear on the line of water-limited potential yield, and the other would be well under the line, assuming a good HI for both.

Another way of interpreting the interaction between canopy development and water loss by direct evaporation is to look at the marginal rate of extra evapotranspiration induced by additional leaf area. This marginal rate can approach 0 if the distribution of rain is such that the soil surface remains wet during much of the crop's vegetative phase, for the evaporative demand by the environment may be met whatever be the ratio of transpiration to direct evaporation.

Agronomy and breeding can both strongly affect canopy development. Poor establishment of seedlings results in slow development, and is an especial problem with the semidwarf wheats that are used widely throughout the world. The difficulty is that these wheats have short coleoptiles that do not extend to the soil surface if the seed is sown more deeply than about 60 mm. The leaf emerging from the still-buried coleoptile is then trapped within the soil and the seedling dies. In well-managed well-structured soils with reliable weather this is no problem because depth of sowing can be precisely controlled. But in semiarid environments, especially where there are soil structural problems and where seed is sown directly into untilled soil, the timing of sowing is often dictated by the weather, and the seed bed is often rough. The imperative for getting the crop sown quickly in an unreliable environment has resulted in changes in agronomic technology during the last 20 years. The sowing machines are large (10–15 m wide) and fast (traveling at speeds of up to 10 km h^{-1}), and the sowing boots, which place the seed in the soil, bounce around a lot. A large proportion of seeds can then be sown too deeply. If the subsoil is wet, moisture-seeking sowing equipment can sometimes deal with this problem, but the valleys created by press wheels can be filled with soil if there is heavy rain, which can make the problem even worse. To solve this problem, wheat breeders are using dwarfing genes that provide the benefits of short stems but do not overly restrict the maximum length of the coleoptile and lead to better establishment (Ellis *et al.*, 2007; Rebetzke *et al.*, 2007).

Even if the emergence of seedlings is good, the rate of development of leaf area may be suboptimal. Leaf growth is strongly affected by the temperature of both air and soil, so sowing winter-growing crops early, when soil and air are still warm, leads to good canopy cover during late autumn and winter with consequently less evaporative losses from the soil surface. Changes in agronomic technology during the last 20 years have enabled

farmers to sow their crops earlier, where appropriate. In addition to the changes in machinery mentioned in the previous paragraph, early sowing is enabled by the use of broad-spectrum herbicides. These are used, often in combination, for killing emerged weeds just before sowing, or after sowing and before emergence. Follow-up weed control is enabled by using selective herbicides, also increasingly in combination, for controlling weeds once the crop has established. Repeated cultivation of the soil to control weeds and to make a fine seed bed is no longer needed. These techniques have greatly improved the timeliness of sowing and have enabled sowing to take place many days, sometimes weeks, earlier than the traditional ones whenever conditions are appropriate to do so. They thereby substantially improve yields in water-limited environments where the weather is often fickle at the start of the growing season and opportunities for sowing are best taken when they arise. Breeders have been stimulated by these agronomic changes to produce cultivars that can be sown early yet still flower at the optimal time. Farmers are now able to tune cultivar to season, by choosing cultivars that are specifically suitable for early, mid, or late starts to the season (Anderson *et al.*, 1996).

While excessive plowing of the soil may lead to problems of soil structure and thence poor establishment, lack of plowing may allow soil particles to coalesce, with consequent problems for root growth even when there is seemingly adequate macroporosity (Cockroft and Olsson, 2000). Direct sowing of crops into unplowed soil often results in slow early leaf growth (Kirkegaard *et al.*, 1994a), an effect that disappears if the soil is sterilized (Simpfendorfer *et al.*, 2001). Evidently the physical and microbiological properties of the soil interact. Inhibitory *Pseudomonas* spp. that build up in the rhizosphere in untilled soil may be the main cause of this effect (Simpfendorfer *et al.*, 2002). The slow growth of roots in untilled soil may be a predisposing factor to infection by these bacteria and can be at least partly overcome by seeding with tines that disturb the soil below the seed (Watt *et al.*, 2003).

These major agronomic effects are matched by potential genetic improvements. Richards *et al.* (2002) have reviewed the prospects for breeding for faster leaf area development in wheat. Selecting for large early leaves has produced breeding lines that develop leaf area twice as fast as standard cultivars in common use. Physiologically, the main trait is the size of the first leaf in the embryo. The second most important trait is the specific leaf area, the more vigorous lines having a larger leaf area per unit biomass. Because of the danger that these new genotypes may lead to excessive vegetative growth (Section 6.2), it is important that this rapid development of the main stem that gives good early ground cover is not accompanied by an excessive proliferation of tillers. The incorporation of a gene for inhibiting tiller development may prevent mid-season canopy development getting out of hand (Richards, 2002), and ensures that

resources are not used in producing unproductive tillers. Rapid early growth of wheat is linked to phasic development, with more rapid shoot growth associated with earliness (Davidson *et al.*, 1990). Rapid early growth may also be due to a slightly greater allocation of assimilates to shoots rather than roots, suggesting that there is an association between rapid above-ground growth, early flowering and reduced root growth (Gómez-Macpherson *et al.*, 1998). Breeding to uncouple these traits is needed to combine the advantages of early flowering and increased root growth. Watt *et al.* (2005) report promising results in combining rapid root and shoot growth.

Thus, the trajectory of leaf area development can strongly influence how effectively a crop may use water (see Section 6.2.) Although rapid early development of leaf area can greatly reduce evaporative losses from the soil surface, effectively enhancing the productive flow of water through the plants, there is a trade-off. Too much water used during vegetative growth may mean too little during late floral development and grain filling. In this respect it is notable that some cultivars yield well despite slow development of the canopy. Establishing an optimum rate of canopy development is an unresolved and active research topic, and progress will require a combination of agronomy and genetics.

4.2.2. Stubble management

Covering the soil surface is the most practical way to reduce E_s. Retained stubble from a previous crop is the most readily available material to do that. Unfortunately stubble retention causes problems that hinder its adoption in many environments. These consist of the mechanical problem of blockages in machinery when crops are sown into heavy stubble, and the biological problems of establishing healthy crops in stubble. The mechanical problems can be managed, at a cost, with machinery that cuts, moves, or avoids the stubble while sowing. Prominent among these technologies is precise guidance, using global positioning systems, which minimizes contact between sowing machinery and stubble by sowing seeds midway between rows of standing stubble (McCallum, 2005). In some cases it is necessary to further reduce contact between machinery and stubble by increasing the row width beyond the optimum for crop growth. This problem in turn could be addressed by new management systems and breeding that enable crops to spread into the rows.

The biological problem of stubble is the reduced yield of crops sown into stubble of the same species. Cook and Haglund (1991) showed that the cause of the yield reduction was stubble-borne diseases rather than toxic exudates. Yield reductions caused by stubble-borne disease offset any benefit from water retention in subhumid environments such as the Pacific Northwest of the United States. However in progressively drier environments in the United States the benefit of the additional water from retaining

stubble exceeds the damage from disease, leading to increased yield (Cook, 1990). Continuous cereals can therefore be sown more safely into retained stubble in semiarid environments, but in subhumid environments it is important to grow different crop species in sequence. Even when this is done there can be poor seedling emergence through heavy stubble, for example, with canola sown into wheat stubble. Bruce *et al.* (2006) investigated this by comparing the effects of wheat stubble with an equivalent amount of plastic drinking straws, which were assumed to be biologically inert. The drinking straws caused a similar reduction in the number and growth of canola seedlings, indicating that the problem of stubble was physical rather than related to disease or toxic exudates.

Plastic film is an alternative to stubble as a mulch to reduce E_s. Zhou *et al.* (2009) showed a large yield increase in maize mulched with plastic film between the rows. The yield increase was due to water harvesting and warmer soil as well as to less soil evaporation. The labor involved in laying out film may be prohibitive for farm crops but the results of Zhou *et al.* (2009) show that the concept has potential. Innovation is needed to develop a more feasible and environmentally sustainable system such as a spray-on film which decomposed after crop maturity.

In resource-poor communities stubble is usually unavailable for mulching because it is needed as a source of fuel or stockfeed. It is unclear how to compare the value of stubble for these purposes with its value when retained on the soil (Giller *et al.*, 2009). For cropping systems in both developed and developing countries, most of the beneficial effects of stubble occur before sowing and most of the deleterious effects are during crop growth. A useful compromise is to encourage stubble retention for as long as possible after harvest, followed by partial or total removal before sowing the next crop.

4.3. Residual soil water at maturity

Early studies on water relations showed that crops extract water from soil until it reaches a suction of about 1.5 MPa, once known as the "permanent wilting point." While this idea may apply for plants growing in pots, it does not reflect the uneven extraction of water from a soil profile when a crop matures in dry conditions. Such crops leave the surface layers drier than those near the bottom of the root zone, presumably because the root density and residence time are greater near the soil surface (Dardanelli *et al.*, 2004). This putative limit is also inconsistent with field observations, which show that some dryland crops do not dry the soil profile to 1.5 MPa suction over the whole of the rooting zone (Jordan and Miller, 1980), and can leave as much as 50 mm of seemingly available water behind (Christopher *et al.*, 2008; Schultz, 1971). Subsoils with high sodicity or high concentrations of salt or boron produce even greater incomplete water uptake (Adcock *et al.*, 2007; Rodriguez *et al.*, 2006).

The reasons for substantial incomplete extraction are poorly understood, even when there are no obvious impediments such as boron toxicity or salinity. Only a very small presence of active roots would suffice if they were uniformly distributed in the deep soil and if flow through the soil were the only limitation (Passioura, 1983). Clumping of roots into preexisting cracks and biopores could require water to move further through the soil to the roots, thereby slowing the uptake if the rate of movement were limiting the uptake, but even then a week or two should suffice for the water wetter than 1.5 MPa suction to be taken up unless the biopores were at least 10 cm apart (Passioura, 1991).

Yet other crops dry the soil profile to suctions greater than 1.5 MPa, for example, a subtropical wheat crop growing almost entirely on stored soil water (Angus *et al.*, 1981). In order to deal with actual experience, Ritchie (1981) proposed a field measurement of the water present in the soil profile when a crop matured in apparent stress as the practical lower limit of soil water content. This approach assumes that this limit is a property of the soil and crop species, and is unaffected by crop management, except that crops with higher N-status can dry the soil slightly more than crops of low-N status.

Angus and van Herwaarden (2001) confirmed that increasing the N-status of crops led to more water extraction, in one case 7 mm, which was equivalent to increasing the soil suction from 3 to 5 MPa at soil depths from 0.3 to 0.8 m in a root zone that extended to 1.5 m. More importantly they showed several examples of wheat growing after break crops that extracted, on average, 31 mm more soil water than wheat growing after wheat. In these examples, most of the additional soil water extraction was from the middle parts of the root zone, which in these cases extended to 1.2–1.6 m (Angus *et al.*, 1991; Kirkegaard *et al.*, 1994b), and there was little evidence of additional extraction from deep in the root zone. It is known that an important effect of break crops on wheat is to reduce the level of root diseases caused by fungal, nematode, and insect pathogens by depriving them of a host for a year (Kirkegaard *et al.*, 2008). The additional soil water extracted by wheat growing after break crops is an important mechanism of the break crop effect (Angus and van Herwaarden, 2001). There are many reports of the effects of break crops on production of wheat and other temperate cereals (Kirkegaard *et al.* 2008) but, apart from those mentioned in this paragraph, none that show that additional extraction of soil water accompanies the increased yield. The effect of management on soil water extraction is probably the least well understood part of the crop water balance. Rotations and break crops are known to influence the yield of summer-growing cereals such as corn (Karlen *et al.*, 1994) but the effect appears to be less than with temperate cereals. Presumably soil water extraction by summer-growing cereals is not as strongly influenced by soil-borne disease as temperate cereals.

The duration of crop growth also affects the amount of water extracted from a soil. In a comparison of tropical annual crop species growing only on residual soil water, Angus *et al.* (1983) showed that water use was approximately proportional to crop duration, which varied from 65 to 103 days. Even though short-duration crops such as mung bean appeared to mature in apparent water deficit, other crops such as peanut continued to extract additional soil water from soil layers that had apparently reached their lower limit. Long duration also explains the ability of perennial forages to extract more soil water than annual crops, as discussed in Section 4.1.

Water extraction from the soil can be limited by chemical and physical constraints. Subsoils that are highly alkaline, sodic, or salty, or which contain high concentrations of boron, are inhospitable to roots, which recover little of the water that drains into them (Rodriguez *et al.*, 2006). Even when roots are present in a soil layer their ability to extract water may be limited by the osmotic pressure of salt (Rengasamy, 2002). Acidic soils present similar constraints to water extraction and in some respects pose an even greater problem in restricting access of roots to water contained in benign subsoil beneath an acid subsurface throttle (Scott *et al.*, 2000).

In summary, soil water extraction by crops is determined not only by soil water content, evaporative demand and soil physical properties, as has been long recognized, but also by the physiological status of the crop. Crops that have root systems that are not impaired by pathogens or chemical toxicities will extract more soil water than crops that suffer one or more of these limitations. Such water can, when used during grain filling, increase grain yield substantially (Section 6.3).

5. Transpiration Efficiency

Transpiration is the loss of water from foliage, and is the inescapable partner of carbon assimilation. At the leaf level, intrinsic TE is the ratio of assimilation to transpiration. It depends on the difference between the concentrations of CO_2 in the air and inside the leaf, and the difference in humidity between air and leaf (Condon *et al.*, 2002). It is difficult to integrate behavior at the leaf level to estimate the TE of producing the above-ground crop biomass. This is partly because of the respiratory losses involved in growth and maintenance of the tissues, but mainly because data on root biomass and the often substantial exudation of carbon compounds from roots is usually lacking. Nevertheless, we can be confident that any improvement in TE at the leaf level is likely to be reflected in an improvement in TE of crop biomass.

In agronomic research the term WUE is in common use by agronomists and farmers in referring to the ratio of grain yield to the amount of water

transpired and sometimes to the ratio of yield to total water use. Here we use TE to denote the ratio of above-ground biomass at harvest to total water transpired, which is separated from E_s as shown in Fig. 3. We use WUE to denote the ratio of grain yield to the total water transpired.

It has been recognized, at least since the time of Briggs and Shantz (1916), and thoroughly elaborated by Tanner and Sinclair (1983), that TE is approximately constant for a single environment and that it increases with atmospheric humidity. Bierhuizen and Slatyer (1965) showed for single leaves that the relationship could be expressed as a rectangular hyperbola, that is, TE is inversely proportional to Δe, where Δe is vapor pressure deficit. Kemanian *et al.* (2005) surveyed 11 field studies that measured growth and transpiration of wheat and barley, and average daytime Δe for the growth period. They showed that the relationship could be more accurately expressed empirically as $TE = 4.9\ \Delta e^{-0.59}$. For $\Delta e = 1$ kPa, typical of the mean value during daylight hours at the time of late stem elongation for cereals in southern Australia, this equation gives TE = 49 kg ha^{-1} mm^{-1}.

The strong effect of Δe on TE leads to a possibility of increasing growth at a crop stage when Δe is lowest (Richards, 1991). This effect is compounded in winter-rainfall environments, where the soil surface is often wet during the winter, and the marginal evapotranspiration rate arising from a larger leaf area may be close to 0 (Asseng *et al.*, 2001a). If however the extra growth results in substantially greater water use before anthesis there may be insufficient water available for the grain-filling phase (Section 6.2). Optimal behavior may therefore be subtle and will vary from season to season and among locations (Condon *et al.*, 2004).

With rising atmospheric CO_2 levels, it is likely that TE will increase in C3 crops. Field experiments with free-air CO_2 enrichment (FACE) have shown substantial improvement in biomass, especially where water is limiting (Wall *et al.*, 2006). With C4 crops such as maize and sorghum, FACE experiments have shown negligible growth responses to elevated CO_2 (Long *et al.*, 2006).

Another environmental factor affecting TE is the proportion of diffuse radiation. Assimilation in diffuse radiation is greater than in an equivalent flux of direct radiation, and the diffuseness of the radiation has little effect on transpiration. The dust haze created by the eruption of Mount Pinatubo resulted in greater global photosynthesis (Gu *et al.*, 2003; Roderick *et al.*, 2001). The analysis of Rodriguez and Sadras (2007) suggested that a 1% increase in the proportion of diffuse radiation would lead to an increase in TE of 0.5 kg ha^{-1} mm^{-1}.

TE is greater for crops with the C4 photosynthetic pathway than those with the C3 pathway, reflecting the much lower CO_2 concentration inside the leaves of C4 plants. Rawson *et al.* (1977) found that the TE for single leaves of sorghum (a C4 species) measured in the light was 2.9 times the TE for wheat (C3) when they were compared at the same humidity.

In a comparison of TE for whole crops growing in the field, four tropical C3 species had a similar value of TE when the seed and vegetative biomass were corrected for the glucose equivalent of synthesis using the approach of Penning de Vries (1975), but TE for C4 species was 2.2 times the value for the C3 species when expressed in the same terms. Loomis and Connor (1992) compared TE for different species and reported that the TE for C4 grasses was 1.9 times the value for C3 grasses after normalization for humidity. TE for C3 and C4 species can also be compared from boundaries of the sort shown in Fig. 3. The estimate of TE for wheat is 50 kg ha^{-1} mm^{-1}, based on a WUE of 20 kg ha^{-1} mm^{-1} for grain and a HI of 0.4 (French and Schultz, 1984; Sadras and Angus, 2006). For maize, Grassini et al (2009) estimated TE to be 57 kg ha^{-1} mm^{-1}, which is surprisingly low, considering that they estimated WUE to be 37 kg grain ha^{-1} mm^{-1}. The ratio of these values implies an HI of 0.65, which is unrealistically high. When the value of boundaries for grain and biomass are reestimated using a common value for the E_s intercept, the estimated value of TE becomes 62 kg ha^{-1} mm^{-1}, consistent with the average value of HI of 0.5 for maize, after consideration of grain moisture (P. Grassini personal communication 2009). The ratio of TE for maize and wheat estimated from the boundaries is 1.24, with no correction for Δe. This ratio suggests that the TE for C4 crops may be not much greater than for C3 crops when both are grown in their preferred environments.

In contrast to the strong atmospheric effects on TE there is no evidence of direct effects of crop management on TE, except where it encourages more growth during times of low VPD. The increased effectiveness of water use from management of fertilizer is due to reduced soil evaporation, not TE, as shown in Fig. 3.

TE at the leaf level depends not only on Δe and radiation, but also on the concentration of CO_2 within the leaves. The lower is this concentration within a leaf the larger is the TE, and, further, the less is the discrimination against the heavy stable isotope of carbon, ^{13}C, during photosynthesis (Condon *et al.*, 2002; Farquhar and Richards, 1984). These two relationships together provide an effective technique, based on isotopic analysis of plant tissue, for estimating average internal CO_2 concentration within leaves, and thence the intrinsic TE. This technique has been used to reveal genotypic variation in TE, and to develop new cultivars with large TE (Rebetzke *et al.*, 2002).

Water stress can affect TE if the stomata close, thereby resulting in lower CO_2 concentrations within the leaves, but the physiology can be complex. Although fast onset of water deficits leads to rapid stomatal closure, slow onset, as typically happens in the field if the soil is drying, may have a greater effect on the development of leaf area, with more gradual effects on stomatal conductance (Condon *et al.*, 1992; Hsiao, 1973). Further, stomatal closure usually

results in hotter leaves, and thence larger Δe, so that the impact of the lower internal concentration of CO_2 on TE may be subtle (Condon *et al.*, 2002).

6. HI: Converting Biomass into Grain

HI, the ratio of grain yield to the above-ground biomass, has increased substantially in many crops during the last several decades. The introduction of dwarfing genes into winter cereals to solve the problem of lodging had the unexpected benefit of increasing the HI. In wheat growing in favorable environments, it increased from a maximum of about 0.33 to more than 0.50. The increase has been associated with a proportional increase in the number of kernels set per m^2, with little change in KW or biomass production (Evans, 1993).

Semiarid agriculture has also benefited from this increase, but because the crops typically mature into increasingly unfavorable conditions of rising temperatures and falling water supply, a good HI is generally about 0.40. There are three main challenges in getting a good HI: getting an optimal proportion of water transpired between vegetative and reproductive requirements; protecting floral fertility from major environmental problems (heat, frost, water deficits); and transferring assimilates stored in the stems to the grain.

6.1. Adapting phasic development to environment

The timing of flowering is the most important physiological trait that plant breeders select for in water-limited environments. There is an optimal flowering time at which there is a good balance between water used during canopy development and water used during seed-set and grain filling (Fischer, 1979). Crops that flower before the optimal time may be at risk from frost damage. Those that flower too late are at increased risk of high temperatures and water deficits that can lead to poor seed set. Further, they may have too little water left to allow for adequate postflowering photosynthesis or time to mobilize soluble carbohydrate accumulated before flowering and transfer it to the grain (Richards, 1991). The early recognition by Nix and Fitzpatrick (1968) of the correlation between grain yield and soil water at anthesis in crops growing largely on stored water was an important precursor to the current understanding in this area.

Breeders have generally produced a range of cultivars that flower close to the optimal time in a given environment. A difficulty is that the optimum can vary with the season, for depending on the patterns of rainfall and temperature, earlier flowering crops may do better in one season, and later flowering ones may do better in another. In addition agronomic variables—sowing time and nitrogen nutrition—can influence flowering

time (van Herwaarden *et al.*, 1996). Thus the choice of cultivar has to be consistent with the likely agronomic operations.

As an average there is little room for further genetic improvement in the timing of flowering, though breeders have been producing slower maturing cultivars that can be sown earlier in the season while still flowering at the optimal time (Anderson *et al.*, 1996; Richards, 2006). Such cultivars allow farmers to capitalize on the flexibility in sowing time that their modern machinery and agronomic techniques enable. Global warming may, over the next few decades, bring the optimal time of flowering earlier. Crops normally respond to higher temperatures by flowering earlier, and breeders will no doubt make any further necessary adjustments to phasic development though selection for response to photoperiod or vernalization.

6.2. Managing vegetative growth

As well as an optimum flowering date for dryland crops, there is an optimum amount of biomass at anthesis for a given seasonal water supply (Fischer 1979). Insufficient growth leads to too few grains to maximize yield, since grain number per unit area is closely related to biomass at anthesis (Fischer, 1985). The common causes of too little vegetative growth are late sowing, deficiency of nutrients, particularly nitrogen, and low plant density. Conversely excessive vegetative growth is commonly caused by early sowing, excess nitrogen and high plant density. In conditions of terminal drought, excess vegetative growth leads to exhaustion of soil water, so that grain yield is limited by the amount of water available for transpiration and grains are pinched (Walter and Barley, 1974). When crop nitrogen status is excessive, yield is also limited by low reserves of soluble carbohydrate, as discussed in Section 6.4. Excessive vegetative growth can also lead to foliar diseases and lodging. Control of these problems is described as Canopy Management (Kitchen *et al.*, 2008), a set of practices that can include delaying addition of nitrogen fertilizers and applying growth regulants to reduce stem height. Growth regulants reduce the lodging risk and may also reduce competition for assimilates between the elongating stem and the developing florets. While they have a place with irrigated and favorable rainfed crops, there is little evidence that their use is justified in rainfed cropping. Grazing vegetative crops with livestock is a more feasible means of reducing excess vegetation and can lead to increased HI (see Section 7.2).

6.3. Effects of water deficits, high temperatures and frost on floral fertility

Water deficits during floral development can severely damage seed set, through pollen sterility or abortion of embryos, or can prematurely end grain filling (Saini and Westgate, 2000). Low water potentials or high

temperatures during pollen mother cell meiosis (about 3 weeks before anthesis in wheat) can induce severe pollen sterility and thence low yields in the cereals even though the later water supply might be good (Saini *et al.*, 1984; Ugarte *et al.*, 2007). Low water potentials around the time of anthesis can induce severe embryo abortion, especially in maize, where abortion can be almost complete. Lesions in carbohydrate metabolism in the floral structures are associated with both pollen sterility and embryo abortion (Boyer and Westgate, 2004; Ji *et al.*, 2010).

Frost at anthesis can lead to severe reductions in seed set in cereals, but the search for resistance to frost damage has been disappointing. In wheat there is some evidence of genetic variation, though ears typically suffer damage at temperatures below about −5 °C. The damage is avoided if the ears remained supercooled (Fuller *et al.*, 2007). Barley is more tolerant than wheat, perhaps because anthesis occurs while the ear is still in the boot, and there is promise that a QTL associated with frost tolerance may prove to be useful (Chen *et al.*, 2009).

These various effects of water deficits on fertility can lead to severe, sometimes complete loss of yield in droughted or frosted grain crops. While total loss is rare, drought-induced infertility can unnecessarily reduce yields in seasons in which there is a reasonable water supply but in which water deficits occur at these especially sensitive times. Progress in overcoming these problems is likely to come from resilient cultivars rather than from new practices (e.g., Castiglioni *et al.*, 2008).

6.4. Carbon balance during grain filling

Crops that have a poor water supply during grain filling may produce a large biomass but be unable to match that with a good HI. Excessive vegetative growth, especially that induced by an oversupply of mineral nitrogen, can worsen water limitations by using too much water before flowering. Crops may set a large number of seeds but be unable to produce enough carbohydrate to fill these adequately, neither from photosynthesis after flowering nor from carbohydrate stored before flowering and available for retranslocation. The result is ''haying-off'', in which the crop senesces prematurely, and its yield responds negatively to nitrogen supply (van Herwaarden *et al.*, 1998b).

Van Herwaarden *et al.* (1998a) have argued further that an oversupply of nitrogen worsens an imbalance in water use by reducing not only the amount of potential photosynthesis during grain filling but also the amount of soluble carbohydrates stored in stems and available for retranslocation to the grain. The shortage of soluble carbohydrate arises because of a negative correlation between nitrogen level and storage carbohydrate, which suggests that when too much nitrogen results in too much vegetative growth, the photosynthate is diverted into structural rather than storage

carbohydrate. Breeders may be able to help solve this problem, for there is useful genetic variation in the ability to store mobilizable carbohydrate in the stems before flowering and during early grain filling (Richards *et al.*, 2002; Ruuska *et al.*, 2006). Farmers can get around this potential problem by applying nitrogen fertilizer tactically, in mid-season, once they have a better idea of how much water their crops are likely to get, rather than applying large amounts at sowing (Angus and Fischer, 1991; Hochman *et al.*, 2009b).

Subsoil water can be especially valuable during grain filling in winter cereals. Depending on the season, every mm used during this time can result in up to 60 kg ha^{-1} of grain in wheat crops (Kirkegaard *et al.*, 2007; Manschadi *et al.*, 2006) about three times the seasonal WUE shown in Fig. 1. The reasons for this large marginal response have not yet been well explored. Intrinsic TE (see Section 5) is likely to be low during this time, owing to increased vapor pressure deficits. However, because little structural carbohydrate is being made, the photosynthate will likely be going directly to temporary storage in the stems during early grain filling or directly to the grains during the fastest period of grain growth. As well, the extra water is likely to lead to longer duration of green leaf area, and more time to mobilize stored carbohydrate and transfer it to the grain.

The value of this subsoil water can be amplified if it contains substantial amounts of nitrate, as it may if nitrogen is in excess to the young crop's requirements, whether from early mineralization or from fertilizer that is leached into the subsoil by the opening rains. While nitrate close to the soil surface can produce excess vegetative growth, nitrate in the subsoil tends to be taken up later in crop development when it selectively increases grain protein (Lotfollahi *et al.*, 1997). The greatest contribution from the subsoil water is in the years of moderately good water supply (Lilley and Kirkegaard, 2007). During severe droughts, there is little such water, and in the years of ample water supply it may not be needed.

7. Yield Improvement by Management

New management practices leading to yield increases can be broadly classified as strategic or tactical (Waggoner, 1968). Strategic management is expected to be effective in almost all seasons and works best where crops are grown with adequate and reliable water supply. Strategic management also applies to dryland crops when it is directed to producing reliable yields in average or dry seasons, as discussed in Section 2.1. Tactical management practices are those that vary between seasons, in response to weather conditions, the general standard of the crop, grain prices and input costs.

7.1. Strategic crop management

Examples of strategic management practices are those that must be decided at or before the time of sowing. Examples, among many, are correction of acute nutrient deficiency and the rate and geometry of seeding. In many dryland environments early sowing is usually one of the most reliable strategies to maximize WUE (Stapper and Harris, 1989), as we allude to, in several sections. In the past a serious delay in sowing has been for weed control, but with the release of herbicide-tolerant crops, farmers are able to sow crops with the confident expectation that weeds can be controlled after emergence. The prospects for glyphosate-tolerant crops are better than for tolerance to other herbicides because evolution of glyphosate resistant weeds is relatively slow.

Another reliable strategy is the correction of acute nutrient deficiency. An example is the occasional application of fertilizer to correct the widespread but previously undetected zinc deficiency of dryland wheat in Turkey (Cakmak *et al*., 1996). The success of this relatively recent development is a reminder of the need for vigilance in detecting previously undiagnosed problems and responding to realignment of management practices in response to new opportunities.

A less reliable management strategy is the best-bet sowing geometry, which in the case of sorghum can consist of rows as wide as 2 m for low yield expectations or as narrow as 0.5 m for high expected yields (Myers and Foale, 1981). In this case wide rows do not produce high yields in favorable seasons.

7.2. Tactical crop management

Tactical or flexible management can be as simple as fertilizer application or as complex as a change in the species of crop in response to weather at the time of seeding. The advantage of flexible systems is that they help to close the yield gap in favorable seasons while trimming costs in unfavorable seasons. Tactical management has the potential to increase yields in favorable seasons with little effect on yields in poor seasons.

The variable responses to inputs reinforce the natural variability of yield in dryland systems, where, even without tactical management, a large proportion of the grain production and an even greater proportion of profit come from a relatively small proportion of seasons. For example, Fischer (2006) presented an example of dryland wheat yields over 40 years at a family farm in south-eastern Australia. Over that period the farm produced 50% of the grain in 33% of years and earned 50% of the gross margin (at 2010 prices and costs) in 20% of years.

A striking example of flexible management is the system of opportunity cropping that has evolved in eastern Australia since the 1950s (Berndt and

White, 1976). This is a subtropical region with extremely variable but summer-dominant rainfall. The traditional cropping system was fallow-wheat or fallow-barley but now also includes sorghum, skip-row cotton, sunflower, and mung bean as summer crops and chickpea and canola as winter crops. With opportunity cropping, farmers maintain a fallow after harvest and sow a crop whenever the available soil water content becomes adequate. In dry periods the fallow may continue for more than 1 year while in wet periods double cropping of summer and winter crops may be possible. With retained stubble and weed control using herbicides, the fallows retain more soil water than the previous cultivated stubbles and the crop water use and productivity over a period of years are greater than the previous inflexible system.

Flexible crop management is also possible in an environment with a defined growing season. In the semiarid tropics, Stewart (1991) proposed a system of response farming which recommended sowing long-duration crop varieties when the onset of the wet season was early and short-duration varieties when it was late. Similar opportunities arise for flexible crop selection in the winter-rainfall regions of southern Australia, where an early autumn "break" signals an opportunity to grow facultative-winter varieties of wheat with longer duration and higher potential yield than spring varieties that are suitable for later sowing (Pugsley, 1983). Long-duration crops also offer the opportunity for grazing by sheep and cattle during the vegetative stage, which provides profitable animal production with little or no reduction in grain yield (Virgona *et al.*, 2006). The reason that grazing does not reduce yield is that, because leaf area is removed by grazing, transpiration is deferred until the reproductive and grain-filling stages, when it is used efficiently for grain production (Angus and van Herwaarden, 2001; Kirkegaard *et al.*, 2007). Even mechanical defoliation during the vegetative stage can lead to increased yield of dryland wheat, also because of deferral of water use until grain filling (Zhu *et al.*, 2004).

In southern Australia, an early break to the season also provides a good opportunity to sow canola and other broadleaf crops. The profitability of canola is comparable with wheat with early sowing but canola yield decreases by about 10% for each week's delay in sowing, compared to about 5% for wheat (Hocking and Stapper, 2001). While early-sown broadleaf crops are profitable in their own right, much of their value in the cropping systems of southern Australia comes from the increase in yield of the following cereal (Angus *et al.*, 2001; Kirkegaard *et al.*, 2008). Furthermore wheat growing after a break crop generally gives greater responses to applied nitrogen fertilizer than wheat after wheat (Angus, 2001). As in many dryland cropping systems, the use of nitrogen fertilizer is risky because excessive N levels in a crop can lead to reduced yield and lower grain quality because of "haying-off" (van Herwaarden *et al.*, 1998b). The risks can be minimized by delaying the decision to apply fertilizer until

the crop has reached the stage of stem elongation, when top dressed nitrogen can be just as efficient as nitrogen applied before sowing (Angus and Fischer, 1991). The advantage of delaying the decision to top dress is in saving the cost of fertilizer and avoiding yield loss by not applying the fertilizer if the season is dry. The later the N is applied the less time there is for adverse weather conditions to produce adverse yield responses, provided that the N is not applied after anthesis, after which the yield response is generally low. The assessment of seasonal conditions can be subjective, or based on measured or modeled soil water (Hochman *et al.*, 2009a). It may be possible to assess future seasonal conditions by considering climate indices such as the El Niño-Southern Oscillation (Anwar *et al.*, 2008) but it is not clear whether such forecasts provide better information than the amount of soil water, or indeed the subjective judgment of farmers. Nevertheless, the combination of break crops to control root disease and tactical topdressing of nitrogen fertilizer was responsible for a large increase in dryland wheat yield in southern Australia in the 1990s (Angus, 2001).

7.3. Spatial variability and zone management

Farmers are well aware that their fields vary spatially in yield, but until recently they did not have the tools for quantifying that variability and possibly managing it to improve the performance of their crops. The new tools are yield monitors and global positioning systems (GPS) on harvesters which can generate yield maps, coupled with geographic information system (GIS) databases that enable such information to be stored electronically and compared across different seasons (McBratney *et al.*, 2005). These data can be combined with other spatially referenced information about permanent features of a field, for example, electromagnetic induction which can indicate salinity or water-holding capacity (Wong and Asseng, 2006) and about conditions of a current crop using satellite imagery (e.g., Batchelor *et al.* (2002)).

Although the pattern of spatial variation may vary across seasons, for example, areas of low elevation may do best in dry years and worst in wet years, there are many cases in which particular areas may yield poorly in most years, with consequently low average WUE (Lawes *et al.*, 2009). Further, economic analysis of such areas often shows that farmers may be losing money by farming them—the returns may be less than the costs of the inputs (Passioura, 2002). When that happens it is likely that the soil in these areas is especially poor and that it would be worthwhile to reduce the level of inputs or even exclude the areas from production. Reducing inputs on part of a field would reduce WUE but probably increase profit, while excluding part of a field from production will increase the average WUE and profitability of the crop. The latter option could also become an

environmental set-aside system if the excluded land were sown to perennial plants that would reduce the risks of erosion and of offsite effects due to the excessive leakage of water and nutrients that is likely to occur under poorly growing crops.

7.4. Diagnostic guide

The large proportion of dryland crops that yield below the water-limited potential represents a challenge for agronomists to diagnose reasons for the shortfall. Figure 5 presents an approach to diagnosis, based on a key developed by van Herwaarden (van Herwaarden and Passioura, 2001); see also (Gomez-Macpherson *et al.*, 2003). The first step is to estimate water use, which includes change in soil water between sowing and harvesting as well as the growing season rainfall (Hochman *et al.*, 2009a), and explore the most effective ways of measuring the change in soil water between sowing and maturity.

If WUE is below the locally recognized benchmark, the next steps are to follow the key, starting with an estimate of the HI. Instead of the time-consuming method of weighing grain and straw, an acceptable estimate can be made by harvesting mature grain-bearing tillers at ground level, placing them on a fulcrum and finding the position along the stem at which they remain horizontal (van Herwaarden and Passioura, 2001). This position is related to HI for wheat as shown in Fig. 6.

Working further through the key shows likely reasons for low WUE. If HI is significantly below 0.4, the limitation lies in the list of management and environmental problems that affect grain filling in the first part of Fig. 5.

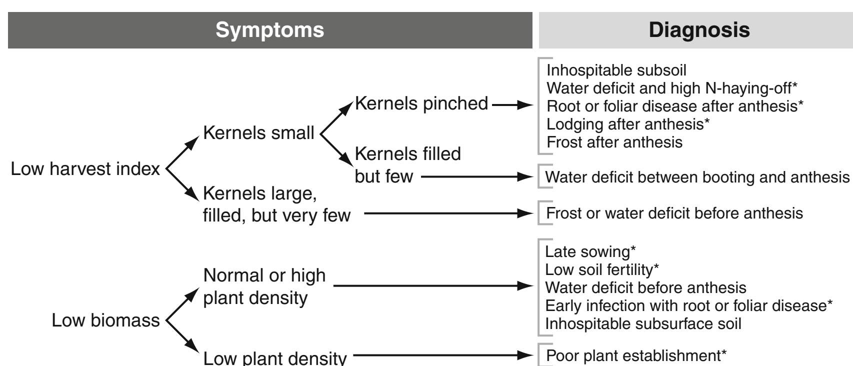


Figure 5 Abbreviated key for diagnosing reasons for low water-use efficiency of dryland crops. Full key available at URL: http://ecoport.org/ep?SearchType=pdb&PdbID=32326 (accessed 27 November 2009). Asterisks (*) indicate problems for which remedies may be possible.

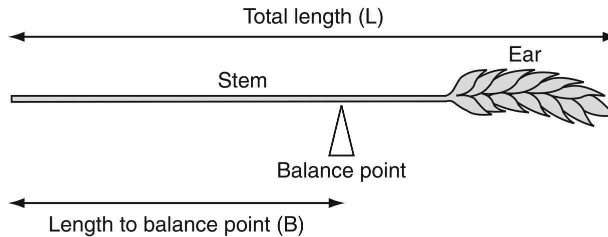


Figure 6 Technique for estimating harvest index of wheat by cutting a stem off at ground level, finding the balancing point, as shown, and measuring distance to balancing point (B) and total length (L). The function relating the ratio B/L to the harvest index, HI, measured by weighing and threshing, is $\mathrm{HI} = 2B/L - 0.96$. Trial and error is needed to find out how many stems need to be measured to get a reasonable estimate of harvest index.

If the HI is satisfactory the limitation is associated with biomass production. Where the limitation is due to management there is scope for improvement in subsequent crops.

7.5. Effects on economy and environment

Rainfed cropping is risky. To minimize risk, conservative and traditional cropping systems manage crops for an average or poor season and forgo the high yields possible in more favorable seasons. Tactical and zone management practices can seek to limit risks but most methods to capture the advantages in favorable seasons still involve some increased risk of loss. Even when tactical management produces higher yields and profit over a period of years, there are problems that have to be overcome. Large amounts of grain produced in favorable seasons require storage and may attract low prices. Further costs may be incurred from storage of seed and fertilizer in anticipation of favorable seasons so the productivity gains should convincingly exceed these costs if tactical management is to be effective. In some cases new machinery, such as high throughput fertilizer spreaders, and new infrastructure such as better transport, storage, and access to markets is needed before the full benefits of tactical management can be realized.

There are likely environmental benefits from greater yields. The resources of water and nutrients that are unused by fallows and low-yielding crops can lead to long-term environmental damage, such as dryland salinity and eutrophication, so in a sense the stability of production in conservative systems may be at the expense of the environment.

8. Concluding Comments

According to measurements of 691 experimental and farm crops of dryland wheat, average yield was 32–44% of the potential yield (Sadras and Angus, 2006) though some of the crops surveyed were grown several decades ago when yields may have been unrepresentative of the current practices. The water-limited potential yield probably represents an unrealistic expectation for commercial crops and the 2008 World Development Report proposes that attainable yield on farms is 80% of potential yield (World Bank, 2008). Using this value, the average yield is about half that attainable on farms. The gap between actual and attainable yield may be due to both management and environmental factors and it is not possible to accurately partition the yield gap between the two. In Australia, part of the management-related yield gap was closed during the 1990s through greater use of break crops to reduce soil-borne disease, and tactical use of nitrogen fertilizer (Angus, 2001). Most wheat crops in Australia and other semiarid wheat-growing regions are not preceded by a break crop, so further increases in regional wheat yields are likely if break crops are adopted more widely. Other management improvements such as earlier sowing can further help to close the yield gap.

The rate of progress towards the attainable yield of dryland wheat has been estimated by Fischer *et al.* (2009) to be less than 1% per year, made up about equally by improved crop management and breeding. At this rate average farm yields will not reach the attainable potential for at least 50 years. This rate of yield improvement is not enough for human nutritional needs. Anderson (2010) questions the relative contributions of crop management and breeding and concludes that improved crop management is responsible for most of the increased productivity.

In the foreseeable future dryland crops will experience a certain increase in atmospheric CO_2, a likely continuing increase in temperature, and uncertain changes in the water balance. As discussed in Section 5 and based on FACE studies, increased CO_2 will increase WUE of C3 crops but not of C4 species. The effects of rising temperature on crops are generally negative for all but the coolest environments, but are subject to adaptation by breeding (see Section 6.1). Possible changes in water supply at a regional level are less easily predicted than CO_2 or temperature (Lim and Roderick, 2009), but if negative, would have more effect on dryland crops than increases in CO_2 or temperature. Even with the worst climate scenarios the rate of change of crop production is likely to be less than that of current yield improvement. However the requirement to improve food security and adapt to several aspects of climate change will require faster improvements in crop management and breeding. This review argues that

productivity of dryland crops can be substantially improved. However, an international commitment to innovation will be needed to achieve the improvement.

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CHAPTER THREE

Advances in Understanding the Molecular Structure of Soil Organic Matter: Implications for Interactions in the Environment

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Abstract

We take a historic approach to explore how concepts of the chemical and physical nature of soil organic matter have evolved over time. We emphasize conceptual and analytical achievements in organic matter research over the last two decades and demonstrate how these developments have led to increased skepticism toward the humification concept and question the usefulness of operationally defined and artificially prepared "humic substances" as models of fractions of soil organic matter in real soils. We identify some of the still open questions about organic matter, and point out future research directions that are likely to refine those concepts of the physicochemical nature of soil organic matter which we put before the reader today.

1. Introduction

The dynamics of soil carbon, and specifically, the question of what determines the length of time an assimilated carbon atom remains in the soil before it is respired and released into the atmosphere, are central to the welfare of mankind for three major reasons. First, there is uncertainty about how the vast amounts of soil carbon are going to respond to climate change (Heimann and Reichstein, 2008). Second, the rising mean temperatures of the atmosphere have prompted attempts to store more organic C in soils for longer times, thereby removing CO_2 from the atmosphere and making the atmosphere cooler or,- at least mitigating further warming (Lal, 2004). Finally, well-balanced levels of soil organic matter (SOM) are necessary to ensure nutrient release and biological activity and thus, the ability of soils to sustain agriculture and to feed the growing human population (Stewart *et al.*, 2007).

The agricultural benefits of organic matter come with the penalty of a mass loss because organic matter is most beneficial, biologically, when it is decomposed by microbial activity (Janzen, 2006). Consequently, managing

the SOM resource in a way that optimizes soil fertility and maximizes agricultural production was the big historic driver for research into the turnover characteristics of SOM. The earliest published experiments on organic matter turnover were carried out by Jethro Tull (1751), who established that frequent tillage with plows could substitute for manure or fallow in continuous cropping of wheat, thereby speeding up mineralization of organic matter through increased aeration (Warkentin, 2000).

1.1. Molecular structure and carbon turnover in soils

What makes us think that organic matter turnover should be linked to its chemical and physical structure? Significant morphological differences between the organic matter enriched layers of different soil types are easily observed and can be empirically related with variations in macroscopic patterns of organic matter accumulation and decay (Feller *et al.*, 2003). From such historic empirical observations, a logical step leads to the hypothesis that the physical and chemical nature of SOM should be a control on the rates of soil carbon cycling. Here we encounter the greatest intellectual challenge that the study of SOM has to offer: the necessity to reconcile the long standing perception that SOM is somehow both "dynamic" and "refractory" at the same time, and the question of how such a duality should be reflected in its chemical and physical properties.

The dynamic nature of SOM is immediately obvious to anybody who chooses to till but not to fertilize and does so for several years. The result will be a rapid depletion of organic matter. However, after some time, the declining organic matter contents will approach a new equilibrium level that is considerably lower than that at the start of the experiment. The most obvious and suggestive way to explain such an observation mechanistically is to assume that some organic materials are very hard to decompose for microorganisms because of some special molecular characteristics. But there are other, equally plausible explanations. Some organic matter may be inaccessible to decomposers because of the aggregated structure of soils, or it may be adsorbed on minerals and thus simultaneously protected against enzymatic attack. Neither of these explanations requires organic matter to have a particularly refractory molecular structure. The first modern review of research on microbial biodegradation listed four major hypotheses to account for the postulated resistance of humus or specific organic components to decomposition (Alexander, 1965): (1) Humus is formed from derivatives of lignin, which are combined with amino, peptide, or protein constituents, the complex serving to protect the aromatic and the nitrogenous components from decomposition. (2) It is a resistant polymer or polycondensate of aromatic molecules, such as the phenolic, quinone, or aromatic amino acid type. (3) The carbonaceous components are rendered less susceptible as a result of their adsorption on surfaces and within the

layers of silicate minerals. (4) Humus is preserved because of the formation of stable complexes between its organic constituents and trivalent cations.

However, in the nineteenth century, when it was first realized that SOM seems to decay in a somewhat contradictory fashion [“the destructibility of this vegetable earth is an undeniable fact,” de Saussure, 1804 as opposed to “humus shows such extraordinary constancy that one can look upon it as indestructible,” Hoppe-Seyler, 1889, quotes are excerpts from Waksman, 1936], the involvement of aggregation and adsorption processes in organic matter turnover were not yet known. Thus the notion that SOM should consist of some parts with a molecular structure that decomposes easily and other parts (i.e., dark amorphous, humus compounds) whose molecular properties render them refractory, could over time evolve into the paradigm that humic substances are the organic materials most resistant to microbial attack (Brady and Weil, 2008). This tacit understanding of SOM represents a piece of foundational information that is taught to undergraduate students in introductory courses in soil science.

1.2. Conceptual approach and objectives

Together with an increasing number of fellow scientists (Guggenberger, 2005; von Lützow *et al.*, 2006; Wershaw, 2000), we feel that traditional soil science curricula may have biased our collective thinking about decomposition of SOM towards an uncritical acceptance of the humification paradigm. This early training has planted in most of us the conviction that there is an enigmatic “refractory” fraction within SOM. We are concerned that reliance on the existence of “inherently stable” fractions within SOM may seriously affect the development of concepts to develop robust models of soil carbon turnover.

Here we approach this issue by taking a historic approach to explore how concepts of the physical nature of SOM have evolved over time. We emphasize conceptual and analytical achievements in organic matter research over the last two decades and demonstrate how these developments have led to increased skepticism towards the usefulness of the humification concept and question the existence of operationally defined and artificially prepared “humic substances” as models of fractions of SOM in real soils. We identify some of the still open questions about organic matter, and anticipate future discoveries and advancements that are likely to refine the concepts of the physicochemical nature of SOM that we put before the reader today. Taking a subjective point of view, we make recommendations for addressing the following:

(a) How should the chemical and physical nature of SOM be conceptualized in and affect future areas of research, such as soil carbon sequestration, calculation of soil carbon residence times and carbon cycle models?

(b) How should the physical nature of SOM be presented in educational settings and educational literature?

2. Soil Is an Aqueous System

Life, that is, the ability of assemblies of molecules to grow, reproduce, process energy and evolve, requires a mechanism to separate individual biochemical reactions from their chaotic surroundings by creating compartments that provide controllable environments for specific biochemical processes. Terrestrial environments are regularly subject to additions of meteoric water, that is, to applications of a polar solvent, which requires the presence of permeable, but hydrophobic skins and cuticles to prevent plant and microbial cell contents from dissolving in the rain. The assembly of the membranes that separate these cellular compartments from the surrounding solvent is an indirect process and involves a feature of the water molecule that is called polarity.

2.1. Water as an agent of structural organization

Electronegativity (EN) is the property of an atom to attract electron density to itself or to withdraw electrons from other atoms. An interesting feature of the water molecule is that it is made of two elements that have a large difference in EN. Oxygen [EN = 3.5] is the second most electronegative element in the universe (next to fluorine [EN = 4.1]), while hydrogen [EN = 2.2] occupies an intermediate position on the EN scale (all EN values according to Allred and Rochow, 1958). As a consequence, the electrons in the water molecule are concentrated in the vicinity of the oxygen, leaving the hydrogen atom depleted in electron density. A molecule with an uneven electron distribution, having regions or poles with different electrostatic character, is called polar (Fig. 1).

Polar molecules that readily dissolve in water are called hydrophilic. Such molecules slip into aqueous solution easily because their partially charged regions attract water molecules as much or more than they attract each other. Apolar molecules made out of elements with uniform EN do not have an uneven electron distribution (i.e., combinations of C [EN = 2.5] and H [EN = 2.2] as in alkanes). Such molecules are said to be hydrophobic. Hydrophobic molecules will not dissolve, and in addition, if immersed in water will find themselves pushed together in lumps, sheets, or globules (i.e., the membranes and micelles that allowed the first primitive cells to have an organized biochemistry), as water molecules try to obey the second law of thermodynamics.

The second law states that energy spontaneously tends to flow only from being concentrated in one place to becoming diffused or dispersed and spread out, not vice versa. Thus, all systems must move towards a state of uniform energy = maximum disorder = maximum entropy. Entropy is a numerical

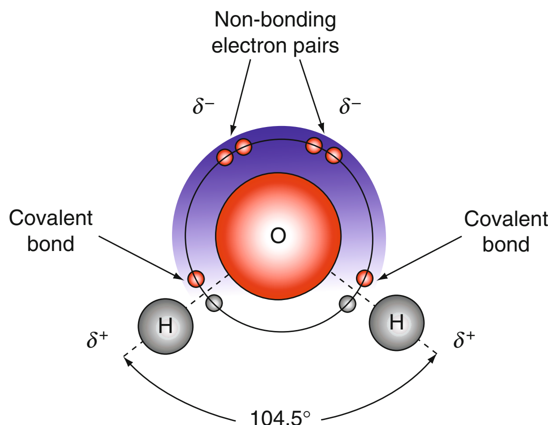


Figure 1 The polar water molecule. Electrons are concentrated around the more electronegative oxygen. Symbols δ^- and δ^+ indicate how partial charge (the ability to exert an attractive or repulsive force) is divided unequally among certain areas of the molecule.

measure of how much this evening-out process has progressed. The second law will always force the components of a system = (water molecules, organic fragments, mineral surfaces) to arrange themselves in a way that achieves maximum entropy. In an aqueous biochemical system containing large and unwieldy biomolecules, the easiest way to achieve this is to increase disorder among the smaller and more mobile water molecules present.

Water is not typically perceived as a particularly well ordered material with a large potential for additional disorder, but this impression is misleading. Liquid water is indeed a relatively well ordered phase owing to the attractive forces between individual water molecules that are called hydrogen bonds. When electron-deprived hydrogen gets to face an area of high electron density on another molecule, for example, an electronegative oxygen or nitrogen molecule, it can form a so-called hydrogen bond (Fig. 2). If it does so with other water molecules, an ordered framework of water molecules is created (Fig. 3).

In the vicinity of hydrophobic organic moieties, water molecules display much slower orientational dynamics than the bulk liquid and therefore have highly reduced mobility, leading to a decrease in entropy (Rezus and Bakker, 2007). Using scaled particle theory, the driving force for the aggregation of apolar molecules through hydrophobic interactions was determined to be the tendency of water to reduce solute cavity surface area (Jackson and Sternberg, 1994), thereby reducing the total number of water molecules in a "frozen" state of low entropy in the solvation shell. Thus the 2nd law of thermodynamics forces nonpolar molecules into clusters reducing the number of surface associated H_2O molecules and maximizing the number of H_2O molecules in the bulk solution (Fig. 4).

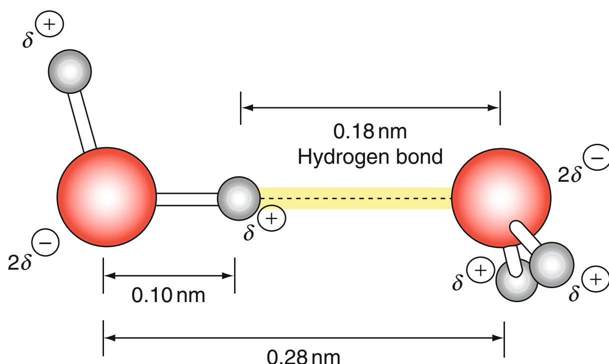


Figure 2 Hydrogen bonds form between polar water molecules. Polar molecules have areas of low electron density (δ^+) and areas of high electron density (δ^-).

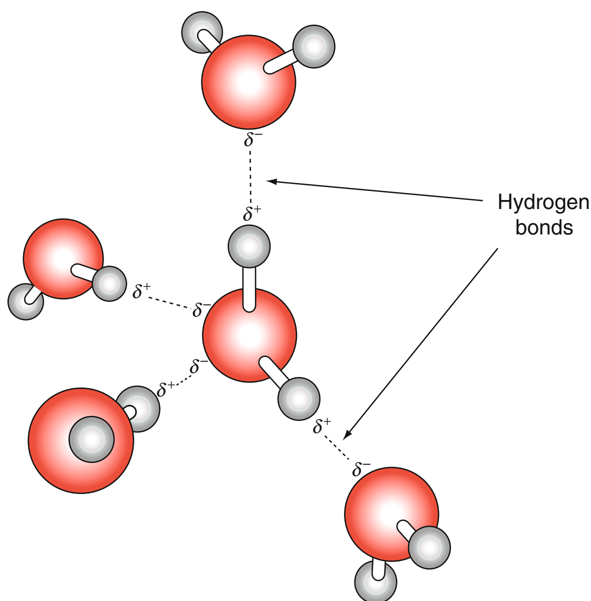


Figure 3 Hydrogen bonds create an ordered framework of water molecules.

The implications of this mechanism for the fate and structure of SOM are far reaching. SOM is defined as the nonliving component of organic matter in soil (Trumbore, 1997) implying that it derives from living plants and animals. All organisms, large and small, simple or complex, share the necessity to separate critical biochemical life processes from interfering external influence by means of membranes, walls, and other structural

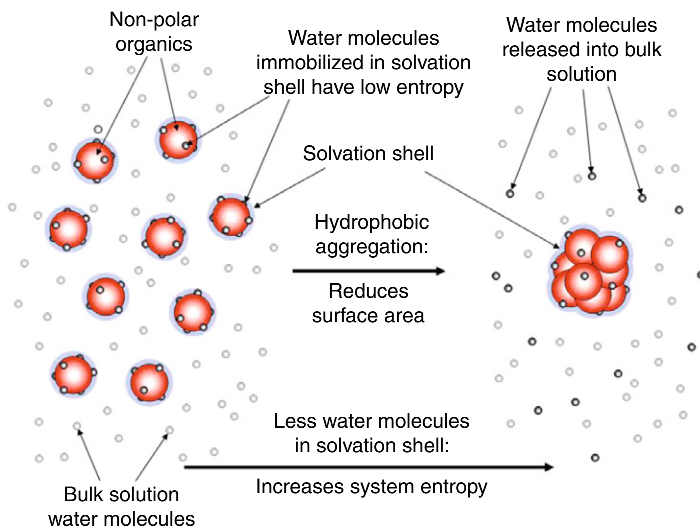


Figure 4 The mechanism of hydrophobic interaction between two apolar organic compounds. Disorder is increased when apolar molecules cluster in an aqueous system. No chemical bond is involved, and the resulting interaction will not persist if the polar solvent is removed.

features that, in order to be functional, must be insoluble in the polar solvent that is ubiquitous on the surface of the earth: liquid water. A quick look out of the window on a rainy day convincingly demonstrates the fact that exterior plant tissues are not soluble in water, and it is easy to demonstrate that the majority of plant litter enters the decomposition cycle in a form that is not soluble in water, that is, as a strongly hydrophobic material.

2.2. The effect of decomposition on aqueous solubility

Decomposers in aerobic soils take advantage of the ubiquity of the second most powerful electron acceptor known in the universe: oxygen [EN = 3.5]. Oxidation of organic compounds proceeds in a way that moves electrons away from the nuclei of reduced carbon atoms in plant materials and closer to the nuclei of oxygen atoms, giving rise to the formation of oxygen-containing functional groups like the hydroxyl (–OH) group or the carboxyl (–COOH) group. The newly added, oxygen -containing functional groups are polar and so convey a partial negative charge in aqueous solution. The carboxyl may even become fully ionized, depending on the pH of the soil solution. Thus these groups confer water-solubility and local electrical charge to the molecules that contain them.

As hydrophobic plant materials enter the decomposition chain, they become increasingly oxidized and acquire greater numbers of

oxygen-containing functional groups, which make them increasingly soluble in water. As a consequence, the decomposing molecule or plant fragment becomes progressively less hydrophobic until it is either completely broken down to CO_2 or has acquired so many polar functional groups that it can dissolve in water. While in that intermediate stage of decreasing hydrophobicity, organic molecules that are partially hydrophobic and partially hydrophilic are called amphiphilic. The fatty acids in cell membranes are often cited as classical examples for amphiphilic molecules while proteins are an example for amphiphilic macromolecules.

To summarize, we note that organic matter in soils may occur in three different forms with respect to their behavior towards a polar solvent:

(a) Some organic materials, like low molecular weight organic acids and monomeric carbohydrates are *hydrophilic* and fully soluble in the polar solvent water. This property greatly facilitates their mobility in the soil environment by diffusion and bulk flow processes and makes them candidates for rapid uptake, mineralization, and leaching.
(b) Structural plant and animal residues tend to be *hydrophobic*, because their biological purpose is to create compartments that allow for the undisturbed progress of numerous different biochemical reactions. If immersed in a polar solvent, these materials will cluster because of entropy driven hydrophobic interactions (hydrophobic interactions: molecules behave like they are trying to minimize contact with polar water molecules). This impedes their mobility within the system and, as it also restricts accessibility to enzymes, slows down decomposition.
(c) Oxygen is abundant in aerobic environments and oxidative decomposition sets in very quickly if moisture and temperature are not limiting. As oxidation is the dominant degradation process for organic matter and as it adds polar/ionic oxygen-containing functional groups to decomposing organic molecules, it is fair to assume that most SOM fragments are rapidly transformed from a pure hydrophobic state and quickly acquire *amphiphilic* properties.

3. Evolution of SOM Concepts

To the eye of the observer, SOM presents itself as a dark, largely structureless substance. From the moment early man took to practicing rudimentary forms of agriculture, it was apparent to those working the soil that this substance was crucial to soil fertility. The origin of the organic matter was equally obvious: observations that can be repeated in any present day vegetable garden show that the recognizable features of organic detritus, either from decaying plant litter or manure applied to the soil converted

over time into darkish, amorphous SOM. Almost 2000 years ago, the Roman writer Columella had recommended "...we may reap greater harvests if the earth is quickened again by frequent, timely, and moderate manuring" (Columella, AD 70). The classic term for the dark, amorphous material that renders soils fertile is the Latin phrase "humus." The dictionary gives the following translations for humus and related Latin words:

humus *-i f. [ground, earth, soil]; humi, [on the ground]; meton., [land, country].*

humilis *-e [on or near the ground, low, shallow]. Transf., of rank, etc., [humble, poor, insignificant]; of character, [abject or submissive]; of language, [mean, without elevation]. Adv. humiliter, [humble, meanly, abjectly].*

humilitas *-atis f. [nearness to the ground, lowness; shallowness]. Transf., [insignificance, obscurity; submissiveness, abjectness].*

Thus, in Latin, the term humus encompasses the whole soil, not just its organic constituents (Manlay *et al.*, 2007).

3.1. Historical approaches to define humus/SOM

The first to publicly recognize humus/SOM as a separate component of the soil system rather than as a particularly fertile "taxonomic" soil type was Wallerius in the year 1761; he defined humus as "decomposed organic matter" (Waksman, 1936), but the prevailing ideas concerning the chemical nature of humus/SOM were still vague at that time. Yet, Wallerius' contribution provided hypotheses that could be tested with the evolving early methods of science in the late eighteenth and early nineteenth centuries. One of these hypotheses was the "humus theory of plant nutrition" (Russell, 1950), which can be seen as an early attempt to develop a mechanistic explanation for observed correlations between high organic matter content and high soil fertility. The carbon of the humus was thought to be an important source of carbon for plants, until Sprengel (Sprengel, 1826) showed that the soluble salts in the humus were the real nutrients (van der Ploeg *et al.*, 1999). By the early nineteenth century, agricultural schools and academies had been established all over Europe, probably motivated not just by scientific curiosity but also by terrible famines like the one that hit in 1816, "the year without summer" (Engvild, 2003). This led to an increase in agricultural research activity and to an increase in opinion among early scientists about the chemical nature of humus/SOM, such that the term "humus" was not always used to designate the same organic substances or preparations (Waksman, 1936). Different conceptions resulted from a wide array of different experimental approaches and observations. For example, it was observed that SOM is not a homogeneous substance; even when collected from greater soil depth it could contain some parts whose recognizable morphological features indicated different

states of processing. Litters from different vegetation types, like grassland versus coniferous forest, were found to produce SOM with distinct and different properties, especially with respect to morphology and acidity. Animal manure was found to be more efficient as a soil amendment when applied after some period of maturation, suggesting its aged or more decomposed state might be physically or chemically different from the raw state. Of particular importance was the fact that the first chemical preparations (i.e., extractions) were attempted in the late eighteenth or early nineteenth century and were found to extract only parts of organic matter, leaving a residue behind that was presumably different from the materials in the extract.

These and similar observations lead to numerous and competing suggestions as to how humus/SOM should best be defined. While some scientists applied the term humus to SOM as a whole, others used it to designate certain fractions of total SOM, depending on their experimental background and observational intentions. A fascinating variety of definitions has been compiled as early as 1936 by Waksman (Waksman, 1936, p. 5ff), and even today there is not yet a unanimously agreed upon definition for humus/SOM within the soils community. Clearly, this lack of a generally accepted, standardized definition is a major obstacle to any attempt to develop a model for the physical and chemical nature of SOM. Even though the problem has been pointed out and explained before in highly visible publications (Baldock and Nelson, 2000; MacCarthy *et al.*, 1990), the issue persists. There are as many concepts and perceptions of humus and SOM as there are scientists and textbooks. We decided to follow the suggestion of Waksman (Waksman, 1936) ''humus should be used to designate the organic matter as a whole'' and use the terms SOM and humus synonymously.

3.2. Alkaline extraction: The procedure that shaped our view of SOM

Few scientific disciplines have been ruled to the same extent by one single chemical preparation procedure as SOM research has been influenced by the alkaline extraction procedure. Over almost two centuries, a bath in a strongly alkaline liquid was one of the first things that happened to every soil sample whose organic matter component was to be studied in detail. It is worth considering how this practice evolved, what it does mechanistically to the constituents of SOM and how its results have influenced our concepts of the physical and chemical nature of SOM.

The manuscript that shaped the traditional models of the physical and chemical nature of humus/SOM was published by Francois Charles Achard (1786), the son of French Huguenot immigrants to Berlin, which at that time was the capital of religiously liberal Prussia. Achard was a disciple of the chemist A. S. Marggraf who had earlier discovered the presence of sugar in

beetroot. Achard's scientific curiosity was unlimited, for a time he was director of the physics class of the Berlin Academy of Sciences, where he investigated all kinds of scientific problems. He developed a method of working platinum and ways to improve the cultivation of tobacco plants in Prussia (Müeller, 2002). At some point the composition of peat excited his curiosity. He reports that the deepest of three peat layers in a deposit near Berlin was the best and the surface layer the worst ("...zum Gebrauch die schlechteste"—least suited for any use). He then gives a detailed account of all the experiments he undertook to investigate differences in the composition of the three peat layers ("...Versuche, die ich gemacht habe um die Bestandteile des Torfes zu untersuchen"—experiments that I made to examine the components of the peat). One gets the impression that he very systematically applied all the different means of extraction, digestion, and preparation that he had available in his laboratory. Among other things he ashed the peat, treated it with HCl, mixed the residue with carbonate and melted the mixture, added sulfuric acid, distilled a peat–water mixture, and quantitatively determined CO_2 evolution upon burning the peat. He even boiled the peat and tasted it, commenting on its bitterness ("...das Ueberbleibsel schmeckte sehr bitter"—the remnant was of a very bitter taste). The10th procedure (out of a total of 14) in his repertoire was the application of a KOH solution, which dissolved a little more than half of the peat in the first layer. The dried residue was much darker than the original peat. Adding sulfuric acid to the extract produced a dark-brown precipitate. Application of an organic solvent (oil of turpentine) led to a slight coloration of the solvent, but did not dissolve a significant amount of the peat. Achard applied this program to all three peat layers, and concluded on the basis of his results that all three consisted of the same basic material ("...das der Torf von allen drey Lagen aus einerlei Grundstoff bestehe"—the peat from all three layers is of the same commodity). After completion of the manuscript, Achard turned his attention to the extraction of sugar from beets and left his peers to deal with three fundamental insights:

(i) Peat does neither dissolve in plain water nor in an organic solvent (turpentine oil).
(ii) Adding H^+ (i.e., strong acid) to the water does not increase the solubility of peat.
(iii) About one half of peat material (40–60%) is soluble when OH^- (i.e., strong base) is added to the system.

The implication of Achard's findings is that organic matter must possess features that influence its solubility in water and are able to respond to base more readily than to acid. These features are commonly called functional groups. Charged functional groups are made by combining a strongly EN, electron attracting element like oxygen [EN = 3.5] with one that has less power to attract an electron, like hydrogen [EN = 2.2]. This creates an

uneven electron distribution within the functional group and makes the molecule polar. Molecules that are, polar are attracted to the unevenly distributed charges within polar water molecule and vice versa. Glucose (Table 1), for example, has abundant polar hydroxyl (OH) groups that can attract water molecules. When crystalline glucose is applied to water, the glucose molecules are attracted to the water and go into solution. Once in solution the molecules stay in solution because they become surrounded by water molecules. A layer of water molecules attracted to another molecule is called a hydration shell (a hydration shell and a water cage differ in that the molecules of a water cage are attracted to each other, but not to the apolar molecule that they surround). A particularly efficient way of attracting water molecules is through functional groups that are not just polar, but ionized. In this case, with the addition of base, a proton has either been lost from or gained by the functional group, which changes the electron imbalance in the functional group to the point where an actual charge is created (Fig. 5). This makes the now charged functional group a much more powerful attractor for oppositely charged (or polarized) molecules, and so ionization of a functional group is a way to massively enhance the ability of an organic compound to attract polar water molecules and become dissolved. And this is what Achard did. His experiment showed that the functional groups in decomposing peat were of a kind that could not be ionized by the application of acid (Fig. 6).

However, the functional groups in peat could be ionized by the application of base. This result allows us to make inference about the nature of the functional groups present in decomposing peat. In order to react with base, these functional parts, or better, functional groups must have the ability to dissociate a proton. The ability to dissociate a proton is called acidity, and it is for this reason that the materials that were extracted by the application of base are traditionally called "acids" (i.e., organic acids). Thus, Achard's experiment demonstrated that organic matter has parts which can develop charge and are able to attract or repel something.

The exact molar strength of the KOH solution used by Achard for his historical first extraction is not known. Contemporary authors (Adani *et al.*, 2009; Bird *et al.*, 2008; Mikutta *et al.*, 2009) use NaOH with concentrations between 0.1 and 0.5 molar, yielding pH values in the order of 13 and above for the extractant ($pH_{NaOH} = 14 + \log[\text{mol l}^{-1}]$). Following extraction with base the procedure calls for the solution to be acidified to pH 1.5. It is illustrative to examine how the most abundant organic functional groups respond to such pH values. A glance at Table 1 shows that raising the pH to a value near 13 ionizes all oxygen-containing functional groups with a p*K*a below 13, which results in greatly enhanced solubility of such molecules. Consequently, such treatment dissolves any organic compound that carries enough oxygen-containing functional groups to create either a continuous water shell around the molecule or whose nonpolar part is small enough to

Table 1 Approximate prevailing dissociation state of functional groups on some organic molecules as a function of pH

	Structure	Functional group	p*K*a	Acidified to pH 1.5	Soil at pH 5.5	Alkaline extract at pH 13
Carbohydrate: Glucose		Alcoholic OH	av. 12	OH	OH	*OH/O*$^{-}$
Alcohol: Ethanol		Alcoholic OH	15	OH	OH	**OH**
Lipid: Pentane		C–H	>50	Nonpolar (not soluble)	Nonpolar (not soluble)	Nonpolar (not soluble)
Lignin derivative: p-hydroxy-benzoic acid		Phenolic OH, carboxyl	9.3 4.5	OH COOH	OH *COOH/COO*$^{-}$	**O**$^{-}$ **COO**$^{-}$
Amino acid: Glycine		Amino Carboxyl	9.6 2.3	**NH_3^+** *COOH/COO*$^{-}$	**NH_3^+** **COO**$^{-}$	NH_2 **COO**$^{-}$
Alkaline extract: Leonardite[a] "humic acid"		Carboxyl I Carboxyl II Amino Phenol	4 6 8 10	COOH COOH **NH_3^+** OH	*COOH/COO*$^{-}$ *COOH/COO*$^{-}$ **NH_3^+** OH	**COO**$^{-}$ **COO**$^{-}$ NH_2 **O**$^{-}$

The pK_a value indicates the pH at which the respective functional group exists as 50% in the ionized form and 50% in its nonionized form. Situations where functional group is predominantly in ionized form (arbitrarily defined as the region where the absolute value of the difference between pH and pKa $\geq |2|$) are indicated by bold type. Intermediate dissociation states (ΔpH-pKa $< |2|$) are indicated as pairs and set in italics. Solubility of a molecule in water increases with the number of polar/ionized functional groups.

[a] Leonardite: yellow to dark brown or rarely black coal that formed from peat at shallow depths and temperatures lower than 100 °C (Westall *et al.*, 1995.)

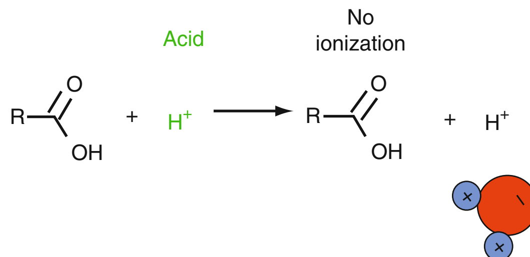


Figure 5 Addition of a proton does not ionize the carboxyl groups in peat. Consequently, the attraction of water molecules operates only through polarity leading to limited solubility of peat in water. *R*, organic moiety attached to the functional group.

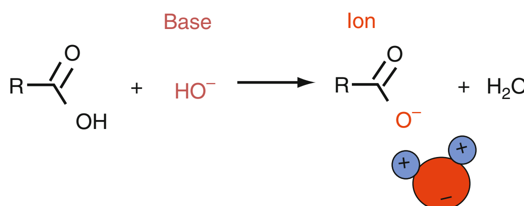


Figure 6 Addition of strong base ionizes the carboxyl groups in peat, creating strong attraction of water molecules contributing to formation of a hydration shell and leading to increased solubility in water. *R*, organic moiety attached to the functional group.

be moved into solution. It also shows that amino groups are not ionized at this pH, making them unresponsive to the treatment. This is one mechanistic explanation for the observation of heterocyclic nitrogen in alkaline extracts, since such nitrogen would be "carried along" as part of organic molecules whose solvation results from ionized oxygen-containing functional groups, while amino N itself is not affected by the extraction procedure.

Table 1 illustrates the principle of alkaline extraction. Nonpolar molecules (e.g., alkyl-C, polycondensed aromatics without substitution) are not extracted by base. Amphiphilic molecules, that is, such that have a nonpolar segment combined with a polar/ionizable segment (like fatty acids), will respond depending on the relative proportions of the nonpolar versus the polar/ionizable part. Some compounds, like ethanol, do not ionize, but are small enough so that the polarity of their alcohol functional group provides solubility in water independent of pH.

With this information we note that the alkaline extraction procedure is more effective at solvating molecules which have oxygen-containing functional groups with p*K*a values below the pH of the extracting solution. Therefore, extraction efficiency depends on number, kind, and quality

(i.e., steric position, p*K*a) of functional groups but is otherwise unspecific with regard to the chemistry of what is on the other side of the functional group, as long as the polar group does not also have to pull into solution a nonpolar section that is too long or too large. The obvious mechanism which increases the proportion of oxygen-containing functional groups on larger biomolecules is aerobic decomposition. Acquisition of O-containing functional groups takes time, which means that more oxidized materials indicate a more advanced stage of decomposition. This is demonstrated by the increased radiocarbon ages of alkaline extracts relative to that of the whole soil (Campbell *et al.*, 1967). However, this apparent increase in age can not necessarily be interpreted as an indication of chemical or thermodynamic stability. On the contrary, the fact that organic compounds are accumulating oxygen-containing functional groups implies that they have undergone decomposition processes. So the apparent "age" of alkaline extracts cannot be used to support the notion that these substances are "refractory." Finally, the alkaline extraction procedure cannot completely extract SOM, because SOM contains organics that are not ionized by alkaline extraction, and consequently do not dissolve.

3.3. Humic substances: True constituents of humus/SOM, artificial preparations or artifacts?

Sprengel (1787–1859) may not have been the first to use the term "humic acids," but he was arguably the one who established it as a scientific term in his work to refute the humus theory (Sprengel, 1826) which assumed that SOM was a major plant nutrient. Soil science continues to struggle with the definitions of the various constituents of SOM. More than 100 years after Sprengel, future Nobel Laureate Selman Waksman attributed the confusion about the chemical and physical nature of humic substances to the fact that scientists eager to apply modern experimental techniques failed to realize that the extracted humic substances are a product of the extracting media and not true constituents of SOM. He writes: "One may feel justified in abandoning without reservation the whole nomenclature of 'humic acids'...beginning with the 'humins'...and ending with...'fulvic acid.' These labels designate, not specific compounds, but merely certain preparations which may have been obtained by specific procedures" (Waksman, 1936, p. 62). With this statement, Waksman implies that humic substances obtained by alkaline extraction are chemically and physically different from the organic materials that actually occur in soil. Additional arguments in support of the notion that the materials created by alkaline extraction (i.e., humic substances) do not exist in this form in soil have been offered by Baldock and Nelson (2000), who point out (i) the highly suspect capability of the alkaline extractable organic matter to represent the true composition

of the whole soil organic fraction, (ii) the apparent lack of a relationship between the biological functioning of organic C in soils and its extractability in alkaline reagents, (iii) differences in the chemical characteristics displayed by the extracted organic molecules when compared to those of the same materials existing in soils in an adsorbed state, and (iv) the creation of artifacts during the extraction procedure (Baldock and Nelson, 2000). This latter point directly relates to Waksman's postulate that the materials represented by alkaline preparations and commonly defined as "humic substances" do not exist in nature. Numerous authors (for historic work see Flaig *et al.*, 1975, p. 25) have pointed out the ability of alkaline extractions to either (i) dissolve not yet degraded plant materials (Zuman and Rupp, 1995, 2006), (ii) induce chemical alterations such as hydrolyses or condensation reactions (Flaig *et al.*, 1975; Head and Zhou, 2000; Tatzber *et al.*, 2007) or (iii) to allow organic materials to become oxidized by air (Swift and Posner, 1972).

In addition, silicate minerals are also known to dissolve at high pH and a variety of alkaline treatments to promote mineral dissolution have been used for this purpose (Hashimoto and Jackson, 1960; Sauer *et al.*, 2006). Thus the alkaline extraction not only modifies the protonation state of oxygen-containing functional groups and cleaves lignin moieties, it also interferes with mineral–organic associations by modifying mineral surface reactivity through dissolution processes, releasing organic matter that was previously adsorbed on mineral surfaces and creating fresh mineral surfaces for the sorption of other organic compounds that have previously been free. To the best of our knowledge, an experimental publication showing that materials extracted with alkali (=humic substances) do indeed occur as such in natural soils has not yet been published. In fact, the opposite is true: Kelleher and Simpson (2006) and Lehmann *et al.* (2008) when using NMR and synchrotron spectroscopy, respectively, were unable to find evidence for the existence of distinct humic molecules in soils.

We conclude that there is good reason for extreme caution when experimental evidence from work with classically defined humic substances is used to infer the physical and chemical nature of SOM. Scientists increasingly recognize this problem and have responded by proposing new definitions to emphasize the physical and chemical difference between artificial alkaline humic substance preparations and natural amorphous decomposition products. Baldock and Nelson (2000) suggested redefining the term "humic substances" to encompass all organic materials that cannot be placed in one of the categories of known biomolecules. An international group of geochemists proposed to call the amorphous decomposing organic materials in terrestrial and marine systems "The molecularly-uncharacterized component of nonliving organic matter in natural environments" or MUC (Hedges *et al.*, 2000).

Many recent publications discussing the physical and chemical structure of humus/SOM (Piccolo, 2001; Sutton and Sposito, 2005; Wershaw, 2004) or mineral–organic interactions (Kleber *et al*., 2007; Mikutta *et al*., 2009; Wershaw, 1993) have used structural and chemical information obtained from alkaline extracts (i.e., humic substances) to make inference about the behavior of humus/SOM as a whole. However, if "humic substances" do not occur in nature, then this practice of using alkaline extractions to characterize SOM becomes problematic and inferences made from alkaline extractions to determine the physical and chemical structure of humus/SOM must be reevaluated.

At this time, the situation is critically unclear. While there is tremendous inertia within the scientific community to keep using the term "humic substances," there is an understanding among the initiated that the materials created by alkaline extractions are highly suspect, as von Lützow *et al*. (2006) stated emphatically: "We abandon terms such as humus, humification and humic substances. . .because these classical definitions derive from traditional operational fractionation procedures that are based on solubility properties." The dilemma is illustrated by an example taken from a recent soil chemistry textbook. Sposito (2008, p. 70) states "Humic substances are organic compounds in humus. . ..The chemical properties of humic substances are often investigated after fractionation of humus based on solubility characteristics." The problem is that (i) if humic substances are the result of various extractions which do not exist in nature, then it is misleading to suggest "humic substances are organic compounds in humus," and (ii) if humic substances are defined as "organic molecules with chemical structures which do not allow them to be placed into discrete categories of biopolymers" (Baldock and Nelson, 2000), then it is not possible to identify their properties based on a isolation procedure which targets only specific functional groups and is known to enhance the breakdown of some of the biomolecules (e.g., lignin). Either way, this is much more than an academic dispute about nomenclature. Scientists use "humic substances" to investigate electron shuttling between metals (e.g., iron, arsenic) and microorganisms implying that these operational alkaline extractions are valid proxies for SOM (Bauer and Kappler, 2009; Jiang *et al*., 2009). Soil carbon turnover models for both agricultural applications and global predictions are based on models like Rothamstead C (Fig. 7), which explicitly considers a "humic" fraction (Jenkinson *et al*., 1991; Kaonga and Coleman, 2008; Liu *et al*., 2009). Most critical are notions of the kind that humification processes might create polymeric macromolecular humic substances which resist decomposition because of their complexity (Brady and Weil, 2008), since such historic ideas form the mechanistic basis for contemporary models of, for example, the temperature sensitivity of organic matter decomposition (Davidson and Janssens, 2006).

Figure 7 Structure of the Rothamsted carbon model. Both DPM and RPM decompose to form CO_2, BIO, and HUM. BIO and HUM both decompose to form more CO_2, BIO, and HUM. DPM, decomposable plant material; RPM, resistant plant material; BIO; microbial biomass; HUM, humified organic matter; IOM, inert organic matter. After Coleman and Jenkinson (1996).

4. The "Humic Dilemma" as an Element of Contemporary SOM Concepts

Today, the many suggestions of how to represent and define SOM appear to converge towards two main approaches: one is functional, while the other attempts to use physical and chemical properties of SOM as defining criteria. These are presented in Table 2. The link between these otherwise quite different approaches is the joint assumption that a suite of biotic or abiotic transformation processes, often summarily designated as "humification pathways," creates the "humus" fraction within SOM which is assumed to be inherently resistant to decomposition, and which is thought to be composed of the "humic substances." The assumed inherent recalcitrance of "humus" has been conjectured from:

(a) Incomplete interpretations of radiocarbon data (e.g., see Table 1 in Falloon and Smith, 2000) where "old" radiocarbon age is interpreted as evidence for the refractory nature of organic matter. However, the fact that an organic compound does not contain much ^{14}C does not necessarily mean that it is chemically and thermodynamically stable. There is no reason why a glucose molecule that is composed exclusively of ^{12}C should decompose more slowly than a glucose molecule that contains one or more ^{14}C atoms.

(b) Structural observations made on alkaline extracts, which, given the evidence presented above, are not representative of SOM because they are chemically altered organic moieties and no longer the *in situ* components of SOM.

In general, these early theories assumed that the "humification" process created polymeric macromolecular humic substances that are resistant to decomposition because of their inherent chemical stability.

Table 2 The two major concepts of soil organic matter (SOM), quoted from Trumbore (1997), Brady and Weil (2008), and Trumbore (2009)

Functional concept	Physicochemical concept
SOM is defined as the nonliving component of organic matter in soil. There is general agreement that SOM contains at least three identifiable C pools: root exudates and rapidly decomposed components of fresh plant litter ("active" pool); stabilized organic matter that persists in soils over several thousands of years ("passive" pool); and a poorly defined "intermediate" or "slow" C pool that has turnover times in the range of years to centuries (Trumbore, 1997, 2009).	"The general term soil organic matter (SOM) encompasses all the organic components of a soil: (1) living biomass, (2) identifiable dead tissues and (3) an amorphous mixture of nonliving tissues, referred to as soil humus. The humus component is divided into two compartments, the (i) chemically identifiable nonhumic substances and the (ii) humic substances" (Brady and Weil, 2008).
Rationale for SOM stability: The ultimate source of organic matter in soils is CO_2 fixed by plants, including leaf litter, roots, and root exudates. The activity of soil fauna (especially fungi and microbial communities) metabolizes some of these substrates and transforms others into more resistant organic compounds which are collectively referred to as humus (Trumbore, 1997).	*Rationale for SOM stability:* Humic substances are characterized by aromatic, ring type structures that include polyphenols (numerous phenolic compounds linked together) and comparable polyquinones, which are even more complex. Because of their complexity, they are considered as the organic materials most resistant to microbial attack (Brady and Weil, 2008).

4.1. Polymeric nature and complexity of "humic substances"

Piccolo (2001) pointed out that the resistance of humic substances to microbial attack has been attributed to its complex, polymeric macromolecular structure, and in fact this view represents information provided in current textbooks (Brady and Weil, 2008, p. 510). Strictly speaking, the terms "macromolecule" and "polymer" define two distinctly different concepts. There are many polymers that are not macromolecules and many large molecules that do not compositionally qualify as polymers of the materials from which they are made. As the original meaning of the term

polymer did not carry any reference to molecular size, the German chemist, Hermann Staudinger, felt it necessary to coin the word "macromolecule" (Staudinger originally used the term "Hochmolekulare Verbindungen"—high molecular weight compounds) in 1922 to describe large covalently bonded organic chain molecules containing more than 10^3 atoms (Jensen, 2008). Today, the official definition (IUPAC, 2006) of the term macromolecule does not include the number of atoms. The IUPAC defines macromolecule (polymer molecule) as "A molecule of high relative molecular mass, the structure of which essentially comprises the multiple repetition of units derived, actually or conceptually, from molecules of low relative molecular mass." Two notes accompany the IUPAC definition:

1. In many cases, especially for synthetic polymers, a molecule can be regarded as having a high relative molecular mass if the addition or removal of one or a few of the units has a negligible effect on the molecular properties. This statement fails in the case of certain macromolecules for which the properties may be critically dependent on fine details of the molecular structure.
2. If a part or the whole of the molecule has a high relative molecular mass and essentially comprises the multiple repetition of units derived, actually or conceptually, from molecules of low relative molecular mass, it may be described as either macromolecular or polymeric, or by polymer used adjectivally.

The reader is advised to keep this definition in mind when following the debate about size and polymeric nature of organic materials in soils (Piccolo, 2001; Sutton and Sposito, 2005; Swift, 1999). According to the IUPAC definition, small molecules can very well be polymers, and a molecule made of many relatively small monomers can qualify as a macromolecule even when its absolute molecular weight is unspectacular. As a consequence, macromolecularity, polymeric nature and high molecular weight are ambiguous arguments to support or reject certain concepts of molecular structure, since the same molecule can in theory be polymeric but have only a modest molecular weight. An analogy between polymers and organic materials in soils that is quite often encountered in publications is the notion of soil organic materials having a "backbone" similar to that of a linear polymer, with functional groups and/or sidechains branching off the backbone (Ellerbrock *et al*., 2005; Mikutta *et al*., 2009).

The concept of a macromolecular structure of humic substances dates back to the work of Kononova (1961), who combined earlier reports of high molecular weight of alkaline humic preparations (Flaig, 1958) with historic observations that simple precursors could be synthesized into humic like materials (Maillard, 1916, 1917) to postulate for the first time that "humic substances are specific high-molecular weight compounds of a complex nature" (Kononova 1961, p. 99). At that time this supposition

seemed to be perfectly reasonable. In 1958, Kononova had participated in the 7th Congress of the International Soil Science Society where Flaig (Flaig, 1958) presented a paper that estimated molecular weights of humic substances at between 30,000 and 50,000 Da with diameters of spherical humic colloids at between 8 and 10 nm. If such evidence is seen in context with the rise of polymer science bringing about the discovery of numerous polymer materials (e.g., Poly Vinyl Chloride in 1927; Polystyrene and Nylon in 1930; Polyethylene in 1941) which all exhibit strong resistance to microbial decomposition, it is easy to see why the idea of ''polymerization processes creating resistant humic macromolecules'' was so convincing at the time. We all know from everyday life experiences that highly polymerized materials are tough to decompose, especially when they are thoroughly crosslinked; contain no, or only few, multiple bonds; and have many aromatic rings and few methylene groups (Mark and Atlas, 1965).

This historic assessment is important because it provides the basis for the continuous quality model of organic matter of Bosatta and Ågren (1999) who state that ''organic matter quality is the number of enzymatic steps required to release as carbon dioxide a carbon atom from an organic compound. The larger the number of steps (=the greater the complexity) the lower is the quality of the carbon atom.'' An immediate consequence of this assumption is therefore that the decomposition rate of low quality substrates (which are complex) will have a stronger temperature dependency than that of high quality substrates, which require only few enzymatic steps for breakdown. This idea forms the mechanistic core of present day concepts of the formation of SOM (Fig. 8) which posit that the temperature sensitivity of SOM to decomposition is greater for more humified, that is, more complex and thus more recalcitrant materials (Davidson and Janssens, 2006).

The above makes clear that the question of complexity, that is, the question whether SOM is at least partially composed of macromolecular, polymeric structures is of great significance for assumptions of the response of soil organic carbon to changing environmental conditions.

We conclude that existing assumptions about molecular complexity of SOM (Bosatta and Ågren, 1999; Davidson and Janssens, 2006) need to be revised because they are based on inference drawn from investigations of unsuitable proxies (alkaline soil extracts; Mao *et al.*, 2001). However we also observe that there is accumulating evidence suggesting the presence of polymeric species in SOM (LeBoeuf and Weber, 2000a,b; Schaumann, 2006a,b). An almost trivial explanation for this may be that nature perpetually creates four major classes of biopolymers (lignin, cellulose, proteins, nucleic acids) and combines them in highly complex structures (lignocelluloses, glycoproteins, glycolipids, cutins, suberins etc), which at any given time are likely to occur in every living soil (Burdon, 2001).

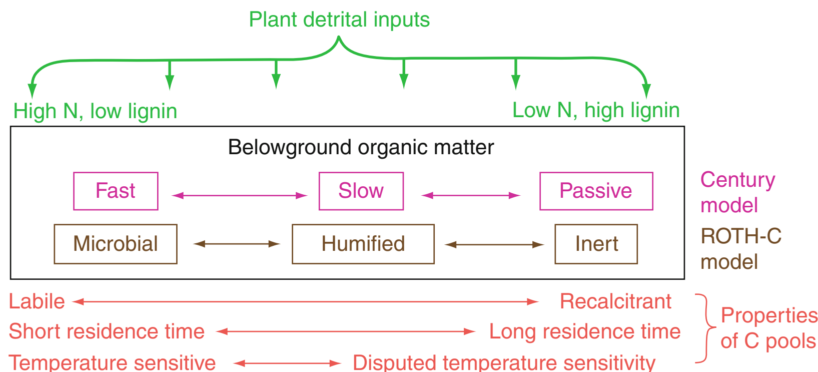


Figure 8 Diagram of properties of conceptual pools of belowground carbon stocks in two well-known models. The CENTURY (Parton *et al.*, 1987) and RothC (Jenkinson, 1990) models each define three discrete soil carbon pools in the mineral soil that lie roughly along a continuum of decomposability and MRT in the soil. The temperature sensitivity of decomposition of the more recalcitrant forms of C is the subject of recent debate. Much of this confusion is due to the fact that the recalcitrant pools are mixtures of simple compounds that have long MRTs owing to physical or chemical protection from decomposition and more complex compounds that have inherently low reactivity and require high activation energy for decomposition. Reprinted by permission from Macmillan Publishers Ltd: Nature, Temperature sensitivity of soil carbon decomposition and feedbacks to climate change, 440 (7081), pp. 165–173, Davidson and Janssens, 2006.

4.2. Aromaticity of SOM

The traditional view of decomposition and "humification" of organic residues in soils included a loss of proteinaceous and carbohydrate materials and an accumulation of "humic" materials containing a high proportion of aromatic carbon (Haider *et al.*, 1975). Today, the view that aromatic compounds accumulate in the course of humification is seriously questioned by many authors (Baldock *et al.*, 1997; Gleixner *et al.*, 2002; Marschner *et al.*, 2008) to the extent that there are doubts regarding the selective preservation of any kind of organic compound (Bol *et al.*, 2009). However, the idea that a humification process concentrates aromatic rings in SOM is still widely accepted, a fact which is reflected in textbook definitions (Brady and Weil, 2008) as well as in published model structures for "humic" materials (Schulten and Leinweber, 2000; Schulten and Schnitzer, 1997a).

We now recognize that aromatic structures are not concentrated and no polycondensate aromatic ring structures are formed during decomposition processes (Baldock *et al.*, 1997; Marschner *et al.*, 2008). A reconsideration of these views from the early 1960s allows us to understand why scientists came to believe that humification processes created recalcitrant molecular structures. The benchmark publication relating aromaticity with turnover time

was presented by Campbell *et al.* (1967). The inference made in this paper was based on light absorption. As alkaline extracts are yellow to brown in color, they must contain conjugated systems of resonating electrons capable of absorbing blue light (Hedges, 1988), and so its dark color is a first indication that SOM may contain appreciable amounts of aromatic compounds.

An early operational procedure introduced by Kononova (1961) measured aromaticity by comparing light absorbance at 465 nm (E4, blue light) with that at 660 nm (E6, red light) and has been suggested to indicate the degree of condensation of aromatic rings, because polycondensed aromatic rings absorb more light at 660 nm than ring systems with a lesser degree of condensation. Campbell *et al.* (1967) noticed an inverse relationship between the E4/E6 ratio and the radiocarbon concentration of alkaline extracts and interpreted these results as an indication that the older, more "humified" material was more highly condensed and more aromatic in nature.

This hypothesis led to the development of two theories that form the basis of present day concepts of the chemical composition of SOM. The first theory postulates that humification processes lead to an accumulation of more condensed and highly aromatic products that are synthesized during decomposition, and that the degree of aromaticity (E4/E6 ratio) provides an index for the extent to which organic matter has been "humified," that is, to which decomposition has progressed. The second theory postulates that humification processes create molecular structures that are recalcitrant and that these recalcitrant molecules are highly resistant to decomposition and may therefore survive for long periods in soils.

Although it is by now clear that aromatic compounds do not experience any kind of selective preservation, making the first theory obsolete (Baldock *et al.*, 1997; Bol *et al.*, 2009; Gleixner *et al.*, 2002; Marschner *et al.*, 2008), the aromaticity of organic materials is still widely used as an index for "humification" (Dieckow *et al.*, 2009; Kalbitz *et al.*, 2000; Ohno, 2002; Ohno *et al.*, 2007; Zsolnay *et al.*, 1999). It is important to acknowledge, however, the fact that state-of-the-art technology has indeed detected high concentrations of aromatic compounds in alkaline extracts (Hatcher *et al.*, 1981; Schnitzer *et al.*, 1991). Examination of the NMR spectra (Hatcher *et al.*, 1981) and X-ray diffraction patterns (Schnitzer *et al.*, 1991) presented by these authors immediately suggests a high likelihood that aromatic structures present in these samples are due to the presence of black carbon (BC), an aromatic product of incomplete combustion and are not the result of natural decomposition processes. While unknown at the time when the first studies on the aromaticity of organic matter were carried out between 1960 and 1980, the evidence suggests that BC is present to some degree in most soils (Haumaier and Zech, 1995; Rodionov *et al.*, 2006; Schmidt *et al.*, 1999; Skjemstad *et al.*, 1996, 2002). Kononova (1961), for example,

proposed the use of humic acid-to-fulvic acid ratios as chemical indicators for the stability of humus. This parameter seemed to be very promising, especially since it could be related to climatic conditions which are known to influence the intensity of decomposition processes. But the vegetation type (Steppe plants with high belowground biomass production) and the climatic conditions (hot and dry summer, cold winter) that lead to the accumulation of thick A-horizons in chernozemic soils also favor episodic vegetation fires with the concomitant production of charred organic matter (Rodionov *et al.*, 2006). This provides an explanation for the high BC contents in grassland soils worldwide (Rodionov *et al.*, 2010) which is completely independent of any humification scenario. Figure 9 illustrates how humic acid-to-fulvic acid ratios in a North–South gradient in Russia appear to reflect climatic conditions for organic matter decomposition (Orlov and Sadovnikova, 2005). Figure 10 illustrates the large amounts of BC in Russian chernozemic soils.

What we observe here is that climate is the likely reason for the concentration of humic acids in chernozemic soils, but not because of its link to "humification" processes, but because summer dryness promotes vegetation fires and BC inputs. The assumption that high humic acid concentrations may convey high stability against decomposition may well

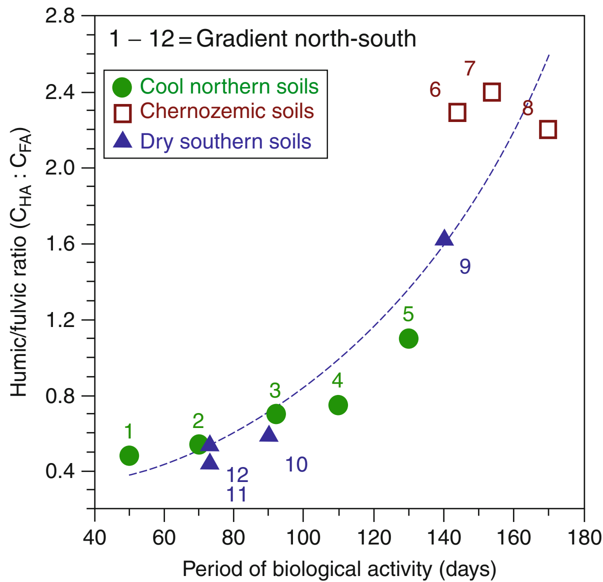


Figure 9 Organic matter composition in a north–south gradient from *Russia* (1 = north; 12 = south) expressed as degree of humification approximated by ratio of carbon in humic acids over carbon in fulvic acids and displayed as a function of biological activity. Data from Table 3 in Orlov and Sadovnikova (2005).

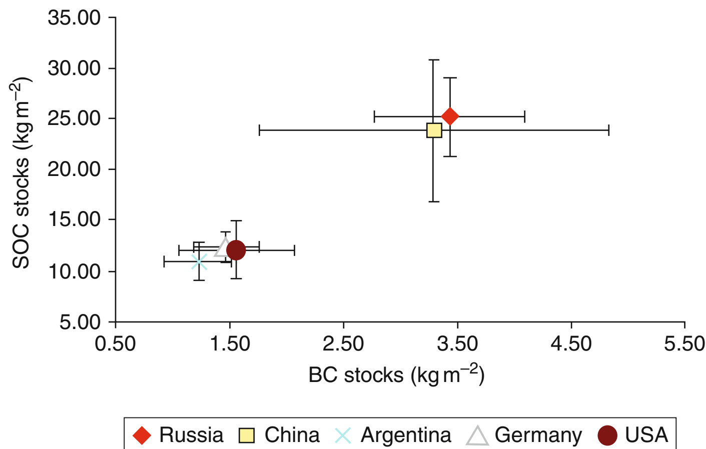


Figure 10 Stocks of soil organic carbon (SOC) and black carbon (BC) in 28 representative soil profiles (mainly Mollisols or Chernozemic soils) from the Russian steppe, the United States Great Plains, the Argentinean Pampa, the Manchurian Plains in China, and the Chernozem region in Central Germany. Figure from Rodionov *et al.*, 2010.

be correct in this case. This is not because secondary syntheses have produced stable humic acids, but because polycondensed BC materials are a low priority substrate for microorganisms.

Publications demonstrating the relative "lability" of lignin-derived aromatic moieties have been around for several decades, and almost every surface soil has at some point seen additions of aromatic BC. Still, large parts of the science community use aromaticity-based humification indices to derive a measure for SOM decomposition state that is then thought of and used as a measure for soil quality. This illustrates a disconnect between those scientific communities working to understand the chemistry of SOM on a mechanistic basis and those that are more concerned with applied questions of land management and have a need for operational metrics.

4.3. Relating the molecular structure of organic matter to carbon turnover dynamics

4.3.1. The paradox of the "stable but dynamic" organic matter

We have shown above that the history of scientific discoveries put the scientific community on the track to focus on some chemical property of organic matter as the explanation for the slow turnover of some organic materials. The logical starting point into any investigation of the origin of such hypothetically stable substances is an examination of the turnover characteristics of the organic plant precursors of SOM. Empirical evidence

from the monitoring of soil A and O horizons over time suggests that the four major groups of organic materials (lignin, cellulose and hemicelluloses, lipids, and proteins) decay without much difficulty when added to the soil as plant litter, although at different rates. A chunk of wood tends to need more time for complete decomposition than a fallen leaf. But examination of any soil profile illustrates (Fig. 11) that even woody plant litters disappear over a period of time while a dark amorphous organic phase appears to persist, if not forever, then at least much longer than the decaying plant fragments.

The general suggestion of this morphological evidence seems to be that of a synthesis process: relatively labile plant litter materials appear to be transformed and resynthesized into something different, a material that has new chemical properties which enable it to withstand decomposition more effectively than the original plant precursor materials. As documented in the texts of Waksman (1936) and Kononova (1961), scientists readily embraced the idea and translated it into scientific questions and hypotheses. Could it be that the stable part of SOM was refractory because one or more processes of secondary syntheses created novel, chemically distinct molecules?



Figure 11 Unaltered plant litters transition into dark, amorphous soil organic matter with depth in an acid forest soil. The litters give the impression that they seem to be disappearing over time, while the amorphous black material appears to be left over. Image by Mark G. Johnson.

Did not Achard's dark alkaline extracts from peat, Braconnot's dark sugar–acid concoctions and Maillard's brown sugar–amine preparations provide overwhelming support for the existence of such a secondary syntheses pathway?

The two most important points of criticism of this approach were raised right from the beginning. Van Bemmelen (1888) concluded that alkaline preparations, that is, the proxies used by scientists to test the hypotheses, were (i) *not true chemical substances* but amorphous and colloidal derivatives of plant substances, and Waksman (1936) maintained that they were (ii) *not particularly stable*, writing on page 87 of his book: "the literature on humus contains statements that humins are highly resistant to decomposition... they were added to soils and their decomposition was measured ... the results point to a complete lack of justification of the assumption." These voices left the scientific community undeterred. Scheffer and Ulrich (1960) defined humic substances as "a first, in the natural environment relatively stable *end product* of organic matter after the termination of life processes" and even contemporary authors (Orlov and Sadovnikova, 2005) maintain that "...thermodynamically and kinetically controlled processes lead to formation of humic acids, representing the *most stable forms* of organic substances. . . ."

Most members of the scientific community were convinced of the plausibility of this rationale and used the process idea of secondary syntheses (or humification) of humic substances as the platform for further research into the structure and properties controlling the turnover of SOM.

4.3.2. Radiocarbon and the paradigm of "what is old must be resistant to decomposition"

While the origins of the paradigm which states that humification processes create molecular structures that are recalcitrant and thus long lived date back to the nineteenth century, the concept received strong support when it became technically possible to determine the concentration of radioactive ^{14}C isotopes in organic materials. Since the rate of radioactive decay is a physical constant, measurements of ^{14}C concentrations allow the observer to calculate the time that has elapsed since the compound in question has ceased to fix CO_2 from the atmosphere. For organisms and parts of organisms, this occurs at the time of death. However, soils experience a continuous stream of organic inputs which have not all died at the same time. The wood in a tree may be hundreds of years old before it enters the soil. In contrast, the photosynthate pumped belowground by a tree to feed symbiontic mycorhizae may have been synthesized just hours ago (i.e., current photosynthate). In general, SOM is a complex mixture of substances spanning an extreme range of molecular properties so that there is every reason to expect that the organic fragments and decomposition products in soils should differ widely in their radiocarbon content. Here the problem arises

that while SOM may actually be a continuum best represented by a distribution of residence times (Trumbore, 1997), continuous distributions are difficult to constrain with field measurements. One strategy to constrain a heterogeneous population is to divide it into subfractions which share enough physical or functional features to allow them to be treated as homogeneous. Among the first to suggest that SOM should be divided into functional pools were Clark and Paul (Clark and Paul, 1970), who proposed that organic components of soil should be viewed as belonging to one of three fractions when considered on a dynamic basis: (1) decomposing plant residues and the associated biomass which turn over at least once every few years; (2) microbial metabolites and cell wall constituents that become stabilized in soil and possess a half-life of 5–25 years; and (3) what they called the resistant fractions, which they thought to be composed of humic components ranging in age from 250 to 2500 years. Their suggestion that SOM contains a fraction that is turning over extremely slow was largely developed on the basis of radiocarbon dating of alkaline extractions, that is, by comparing a mean ^{14}C age for the respective alkaline extract with its aromaticity-based humification index (Campbell *et al.*, 1967).

The observation of old age fit so well with the dominant theory of the stable humic substances that the scientific community as a whole never challenged the conclusion of "what is old is old because it is molecularly recalcitrant." What the scientific community seems to have overlooked at the beginning of the radiocarbon era is that the amount of radiocarbon in a molecule is not mechanistically linked to decomposition. A decomposer organism is indifferent to the ^{14}C content of an organic molecule, meaning that the concentration of ^{14}C within an organic molecule is not at all related to its chemical and thermodynamical stability. In other words, a strand of cellulose containing some ^{14}C has the same probability to become decomposed as a similar piece of cellulose that is exclusively made of ^{12}C, but if dated by the radiocarbon method, the one that has some ^{14}C would appear to be young while the one made exclusively from ^{12}C would appear to be old. So when Campbell *et al.* (1967) postulated the existence of a resistant fraction of organic matter in soils, they unintentionally went a step too far on two levels. First, they used an operational proxy (alkaline humic extractions) which does not exist in soils in similar form. Second, they assumed that low radiocarbon content was the consequence of molecular recalcitrance brought about by a certain molecular property, namely high aromaticity. Such was their confidence in the plausibility of their inference that it never occurred to the authors to demonstrate that their preparations were actually able to withstand decomposition. They merely embraced the prevailing notion of the thermodynamically stable humus (Kononova, 1961; Scheffer and Ulrich, 1960) and concluded that what is old must be refractory, and the mechanistic reason for the alleged resistance to decomposition was taken to be the aromaticity of the material.

This had enormous implications. The concept of functional fractions of organic matter was so appealing that it was adapted as the platform for the first soil carbon turnover models which were developed when computers started to become a readily available tool in science laboratories (Jenkinson and Rayner, 1977; van Veen and Paul, 1981). The idea of a molecularly resistant fraction continues to thrive in the leading present day simulation models (Falloon and Smith, 2000) and can be considered as the foundation of our current understanding of the dynamic nature of SOM (Torn *et al.*, 2009; Trumbore, 2009) and its application to C cycle modeling.

Unfortunately, the automatic assumption that "what is old is old because it is refractory" has never been subjected to rigorous testing. Today, the ability of alkaline preparations to withstand decomposition in soils has been tested with the result that their turnover time is less than decadal (Qualls *et al.*, 2003; Tatzber *et al.*, 2009). Similarly, there is accumulating evidence that "old" SOM (by radiocarbon) may well contain significant proportions of chemically and thermodynamically labile carbon (Sollins *et al.*, 2006). Decomposition does not necessarily lead to increased aromaticity (Baldock *et al.*, 1997). But most important for an updated concept of the dynamic nature of SOM is the realization that accessibility and sorptive interactions with mineral surfaces may provide powerful protection against decomposition, explicitly including carbohydrates, proteins, and other supposedly "labile" materials (Buurman *et al.*, 2007; Derrien *et al.*, 2006; Kiem and Koegel-Knabner, 2003; Kleber *et al.*, 2005; Mikutta *et al.*, 2006, 2007; Rasmussen *et al.*, 2005, 2006, 2007; Spielvogel *et al.*, 2008; Wattel-Koekkoek and Buurman, 2004; Wattel-Koekkoek *et al.*, 2003).

4.4. Model humic substances as proxies for SOM

"The earlier investigators believed that humus formation takes place in nature by a specific process of 'humification' whereby the plant residues are bodily transformed into dark colored substances, or humus" (Waksman, 1936). It is obvious that research into the existence of biotic or abiotic "humification" processes only makes sense if humic substances are entities that do indeed exist naturally in soil. Thus the basis of research into the humification concept is the assumption that a humic substance such as a humic acid is an identifiable entity which possesses some vaguely defined general chemical structure which is created by one or more biochemical humification pathways.

Although Waksman (Waksman, 1936) and others did their best to point out the operational nature of alkaline extracts and preparations, the bulk of the scientific community followed the reasoning of Kononowa, who stated in her classic textbook "At the present time there are no grounds for doubting that humic substances are high-molecular-weight compounds" (Kononova, 1961), p. 84. Consequently, during the recent prime period of

research into humification processes between 1960 and 1990, humic substances were not viewed as artificial products formed during the harsh treatment of the soil with alkali and strong acid, but were believed to be naturally occurring, identifiable chemical substances with the following signature properties: (i) dark color indicative of high aromaticity (Hatcher *et al.*, 1981; Schnitzer *et al.*, 1991), (ii) high molecular weight and (iii) inherent resistance to decomposition (Aiken *et al.*, 1985). Stimulated by the theories put forward by Kononova, a large international effort to elucidate the biochemistry of humification gained momentum in the early 1960s and produced a huge literature (see reviews by Haider *et al.*, 1975; Huang, 2000; Stott and Martin, 1990) on methods to synthesize dark colored, polymeric materials, which typically decompose somewhat slower than their monomeric precursor materials (Tables 3 and 4, p. 48 in Stott and Martin, 1990). The fact that scientists managed in laboratory settings to reproduce some of the characteristics that they observed in the artificial humic preparations, was taken as evidence that they had been successful in reproducing the mechanisms operating in nature. Such was the level of confidence within the scientific community that there was virtually no criticism during this period regarding the appropriateness of using extraction artifacts (operationally defined as humic substances) as proxies for the reconstruction of natural processes. Even today, uncritical confidence in the usefulness of alkaline extractions for research into carbon turnover processes in soils is still widespread within the scientific community (Adani *et al.*, 2009; Pinton *et al.*, 2009).

Attempts to recreate humic substances and humification processes in the laboratory can be divided into three strategies, including (i) treatment of organic precursor materials with acid or alkali (Braconnot, 1819; Maillard, 1916), which yields a dark precipitate that is soluble in alkali. This process is considered to be too artificial to represent natural soil processes, but did influence some of the early theories of humification. More realistic possibilities for a chemical modification of organic materials into products of secondary synthesis are (ii) enzymatic/biotic syntheses reactions and (iii) synthesis reactions catalyzed by mineral surfaces.

The experimental setup for the investigation of these synthesis reactions is straightforward: Following the assumption that humic substances are polymeric, aromatic and recalcitrant, scientists have created laboratory systems that were likely to lead to products with the above characteristics. The early work focused on enzymatically-catalyzed reactions, with the assumption that humification is largely a microbially mediated process. Excellent reviews of this early work are provided by Flaig *et al.* (1975) and Haider *et al.* (1975). Typically, phenolic monomers are dissolved in a phosphate buffer adjusted to near neutral pH and an oxidizing enzyme is added, most often polyphenoloxidases or tyrosinases (Haider and Martin, 1975; Naidja *et al.*, 1998). A multitude of laboratory experiments following

the same basic scheme have been carried out and continue to be conducted (Ahn *et al.*, 2006), recently with a more applied focus on the transformation of phenolic pollutants (Gianfreda *et al.*, 2006; Iamarino *et al.*, 2009).

Polyphenoloxidases naturally occur in plants and microbiota. There is, however, no information about how a decomposer organism would benefit from the large expenditure of resources (e.g., energy, carbon, and nitrogen) to make the enzyme for the purpose of synthesizing a recalcitrant polymer. Activities of free polyphenoloxidases in the soil solution are probably negligible because these enzymes are short lived (Nannipieri and Landi, 2000). Thus, enzymatic catalyses of polymers in the soil environment have a plausibility problem: How would a decomposer organism benefit from developing the strategy of making enzymes which then catalyze the formation of recalcitrant macromolecules? The alternative view would be that catalyzing enzymes are made for purposes unrelated to the transformation of organic matter. They would be inadvertently released into the environment to engage in humification in a random and accidental fashion. We find that neither option is very convincing.

Similar concerns apply to the effects of abiotic catalysis (Huang and Hardie, 2009). While soil minerals with the ability to catalyze polymerization reactions are theoretically ubiquitous, they need to be accessible within the three-dimensional pore structures of the soils and possess active redox domains. The latter is a function of crystallinity, with freshly precipitated and thus poorly crystalline minerals typically having much higher surface site densities than well crystalline, aged minerals (Cornell and Schwertmann, 1996). Laboratory experiments of abiotic catalysis are typically carried out in aqueous phase, such that the probabilities of interaction between reactants and catalyst are maximized, and reaction kinetics boosted by a pH in the neutral region and an elevated reaction temperature compared to standard soil conditions (Jokic *et al.*, 2004b). It is worth mentioning that many trihydroxyphenols and benzoic acids undergo autoxidative reactions at pH values of about 6.0 and higher to form quinones which link with other phenols, phenol polymers, amino acids or peptides to form polymers (Giannakopoulos *et al.*, 2009; Martin and Haider, 1971). However, most soils in active weathering environments develop towards pH values well below those that appear to be required for the polymerization reactions mentioned above.

Another important difference between biotic and abiotic catalyses is that in enzymatically catalyzed reactions O_2 functions as an electron acceptor, whereas in mineral catalyzed reactions, the catalytic minerals themselves (often pedogenic oxides or phyllosilicates containing oxidized iron) function as the electron acceptor, with subsequent loss of catalytic efficiency because the mineral catalyst is being used up in the course of the reaction, while atmospheric oxygen supply is unlimited. Finally, in abiotic catalyses, adsorption of reaction products on mineral surfaces can quickly limit the

extent of polycondensation, favoring the formation of components with lower degrees of aromatic ring condensation and lower molecular weights compared with those generated in the presence of enzymes (tyrosinase, Naidja *et al.*, 1998).

We conclude that attempts to recreate humic materials in the laboratory are flawed by a fundamental misconception. What happens in nature is not adequately represented by alkaline preparations, yet the properties of these alkaline preparations, mainly high aromaticity, and macromolecularity have been used to guide syntheses experiments.

The way forward is to (i) avoid interfering extraction procedures, (ii) investigate the molecular nature of decomposing organic matter with non-destructive techniques, (iii) evaluate if there is evidence for any unique synthesis products, and (iv) attempt to determine the metabolic pathways leading to the formation of such products.

We started this chapter with a quote from the book of Waksman (1936). He also found the right words to close it: "...Humus is not an intermediate product of decomposition.... Neither is humus a final product of decomposition, as was believed by some chemists, who attempted to establish the chemical nature of this final product by its extraction with alkalis and precipitation with acids" (Waksman, 1936, p. 88).

4.5. Organic nitrogen: N-Heterocycles as indicators for humification processes

When soil samples are subjected to an extraction procedure like acid hydrolysis, a nonhydrolyzable residue is obtained that contains appreciable amounts of nitrogen (Janel *et al.*, 1979; Plante *et al.*, 2006). The resistance of some of the organic N to the relatively brutal acid hydrolyses procedure can be viewed in conjunction with the observation that only about 1–3 % of organic nitrogen is mineralized on an annual base (Bremner, 1967). The assumption can be made that the two observations are causally related and that there may be forms of nitrogen in soils that are particularly stable against both chemical and biological attack because they have a particular molecular structure which renders them nonhydrolyzable. However, such an assumption is not automatically plausible because the organic nitrogen compounds in plant litters and faunal and microbial remains (amino sugars, proteins, nucleic acids, etc) do not have a reputation for being particularly resistant to decomposition. Thus, to support a causal relationship between resistance to hydrolysis and slow turnover, one or more mechanisms need to be in place which can explain the formation of recalcitrant organic N from the rather labile precursors predominant in soils.

A variety of mechanisms have been proposed to explain the perceived stability of organic nitrogen in soils, including physical inaccessibility in pores, adsorption to mineral surfaces, addition reactions of inorganic N to

other organic molecules (Stevenson, 1982). Of particular interest in this context is the suggestion that labile precursor molecules might be subject to chemical alterations into more recalcitrant forms through polymerization reactions with other organic soil constituents (Bremner, 1967). The latter option was a popular candidate for a mechanistic explanation to support a relationship between chemical properties and slow turnover of organic nitrogen since the times when it was first noticed that the dark colored nitrogenous substances formed by such reactions showed similarity with alkaline extracts (Braconnot, 1819; Flaig *et al.*, 1975; Kononova, 1961). This theory received support in the 1960s when it was observed that some of the nitrogen in polymers formed by reaction of aromatic monomers with amino acids was not dissolved by the acid hydrolysis procedure (Bremner, 1967). Admittedly, this seemed to fit seamlessly with one of the oldest concepts for the formation of humic substances, the ''sugar–amine theory'' involving the Maillard reaction. In this pathway, monosaccharides and amino compounds recombine via abiotic condensation reactions which result in the formation of brown nitrogeneous polymers or melanoidins that contain pyrrole, pyridine, and other N-heterocyclic residues (Essington, 2003).

Again, we observe that scientists relied heavily on artificial alkaline extractions when attempting to investigate the details of the postulated mechanism. N in heterocyclic molecules was thought to be the product of a humification process, especially the sugar–amine pathway involving the Maillard reaction. Thus, when heterocyclic nitrogen was observed, its existence was taken as evidence that a humification process of the Maillard type was actually operating and transforming labile organic N precursors into a stable, N-containing humic substance. Modern examples for this line of thought can be found in the work of Vairavamurthy and Wang (2002) and Mahieu *et al.* (2000), which both based their findings and conclusions on the examination of alkaline extracts, but not on the investigation of natural soil samples. However, there are a number of indications that the presence of N-heterocycles in soils should be viewed independent of any humification theory.

First, in contrast to the original concept, nonhydrolyzable nitrogen is not resistant to decomposition, as Schulten and Schnitzer (1997b) report. Nonhydrolysable N was found to be reduced to ammonia by biological activity by Ivarson and Schnitzer (1979). Additional evidence for the biodegradability of nonhydrolyzable N has been reported by Keeney and Bremner (1964), Meints and Peterson (1977), Ottow (1978) and Zhuo *et al.* (1995). Mild chemical oxidation converted up to 59% of the nonhydrolyzable N fractions from several humic materials to NH_3 and other N gasses (Schnitzer and Hindle, 1980), further illustrating that this form of N is readily decomposable.

Second, the suggestion that the Maillard reaction might play a significant role in the formation of recalcitrant polymeric macromolecules in soils has been thoroughly discussed by Burdon (Burdon, 2001). Burdon found that the high sensitivity of the Maillard reaction to specific reaction conditions made it highly improbable for this scheme to be a relevant process in most soils.

Third, there is increasing evidence that the N-heterocycles that were identified in many soils are mostly of pyrogenic origin (Almendros *et al.*, 2003; Knicker, 2004, 2007; Knicker et al., 2005, 2008). Other potential sources for N-heterocycles are the purine bases in nucleic acids, and secondary plant metabolites synthesized for a variety of purposes (Leinweber *et al.*, 2009). Neither of these sources is in any way related to traditional ‘‘humification’’ processes. The fact that N-heterocycles tend to be enriched in the smaller size fractions of soils (Mertz *et al.*, 2005) where they appear to be rather resilient over time suggests that their preservation in soil environments is likely a matter of protection in mineral–organic interactions.

We summarize that the proposed sugar–amine syntheses pathway for the formation of heterocyclic humic substances has been convincingly refuted (Burdon, 2001). The alleged inherent stability of natural N-heterocycles (e.g., purine bases etc.) against decomposition has never been conclusively demonstrated, while several sources report on the biodegradability of non-hydrolyzable N. Pyrogenic heterocyclic N within charred organic matter will experience slow turnover due to the specific properties of the BC it is a constituent of. This effect, together with protection in mineral–organic interactions, provides a satisfactory explanation for the observed slow turnover of N-heterocycles in soils. There is no necessity to invoke humification pathways and enigmatic humic substances to justify the existence of N-heterocycles in soils. The presence of N-heterocycles in artificial humic substances is very likely linked to the fact that the alkaline extraction procedure does not ionize amine groups, but obviously ionizes OH and COOH groups that may be part of partially decomposed and oxidized N-heterocycles from nucleic acids and other biomolecules. Thus heterocyclic N embedded in larger molecules with abundant oxygen-containing functional groups has a higher probability to become extracted than the nitrogen in amino groups (Table 1).

The views presented above are at variance with the position of some scientists (Huang and Hardie, 2009; Jokic *et al.*, 2004a–c) who posit that there is a large volume of work documenting the polycondensation and polymerization of simple biomolecules, as catalyzed by enzymes and soil minerals, leading to the formation of humified substances. Since these catalysts are ubiquitous and the substrate biomolecules are readily available, it has been claimed that these reactions occur in the natural environment (Huang and Hardie, 2009). We encourage the reader to take a comparative look at the studies presented by Burdon (2001) and Jokic *et al.* (2004b).

Burdon shows that the Maillard reaction is favored in alkaline systems that contain high concentrations of potential reactants. However, as discussed by Essington (2003), alkaline soils do not contain greater quantities of humic substances than acidic soils, and the concentrations of free sugars and amino acids in soils are exceedingly low in general. Jokic *et al.* (2004b), however, reacted a 0.05-molar solution of glucose and glycine in the presence of 2.5 grams of a manganese oxide catalyst at 45 °C in an oscillating water bath for 15 days, with the pH adjusted to 7. After this treatment, an alkaline extraction was performed and a fulvic acid was isolated. This fulvic acid was shown to contain heterocyclic nitrogen and the conclusion was drawn that a significant fraction of the organic N component of surficial reservoirs may be formed by abiotic, mineral catalysed condensation of sugars and amino acids. We respectfully suggest that this claim would be much easier to accept had the experiment been performed at a temperature of 10 °C, at ambient concentrations of glucose and glycine, in an aggregated porous medium with a moisture content at or below field capacity, and the newly synthesized heterocyclic nitrogen identified in the native organic matter and not in an alkaline extract.

4.6. Molecular structure: Supramolecular assembly or macromolecule?

In the early 1970s, at a time when the assumption that soil humus consists of alkali extractable macromolecules was yet unchallenged, the scientists in search of the general chemical structure of humic materials started to notice that humic substances showed a surprising flexibility in shape and size when exposed to variations in pH and cation concentration. For example, Wershaw and Pinckney (1971, 1973) used small angle X-ray scattering to study the aggregation of soil humic acid fractions as a function of pH, noting both aggregation and disaggregation phenomena when pH was altered. MacCarthy and Mark (1976) and Ryan and Weber (1982) were among the first to show how the addition of metal cations to dilute fulvic acid increased aggregation. While some scientists attempted to explain such dynamic behavior as the consequence of a "flexible coil" structure of the humic macromolecule (Essington, 2003), the realization started to gain ground that "humic substances are not single molecules but rather associations of molecules of origins" (Schnitzer, 1978, p. 46).

4.6.1. The theory of intermolecular associations

The realization that humic substances are not single molecules but rather associations of molecules coincided with the advent of a new branch of chemistry, which provided the theoretical framework to explain the existence of relatively large molecular structures composed of many small noncovalently bonded subunits. In 1987, the Nobel prize for chemistry

was awarded to D.J. Cram, J.-M. Lehn and C. J. Pedersen for their development of what is today one of the most active and expanding fields of chemical research, and for which Lehn coined the term "supramolecular chemistry." Supramolecular chemistry is the science of molecular assemblies; it describes organized entities of higher complexity that result from the association of two or more chemical species held together by mostly non-covalent intermolecular forces. If a disordered system of preexisting components forms an organized structure or pattern as a consequence of specific, local interactions among the components themselves, it is said to be self-assembling (Lehn, 1995). The driving force for self-assembly is the optimization of maximal entropy by dispersing energy throughout the system and so to obey the second law of thermodynamics. The first to apply the idea of self-assembly of the organic materials dispersed within the aqueous soil solution into micelle-like supramolecular structures was Wershaw (Wershaw *et al.*, 1977), who later refined this early idea into a fully developed molecular model (Wershaw, 1986, 1993, 1994, 1999) including a detailed structure for mineral–organic associations (Wershaw *et al.*, 1995, 1996a,b). The rationale of Wershaw's suggestion to treat organic matter fragments in soils as amphiphiles (Fig. 12) assembled into supramolecular structures (Fig. 13) is as follows (Wershaw, 1999): organic materials in soils are formed mainly by enzymatic depolymerization and oxidation of plant biopolymers. These reactions transform the originally nonpolar aromatic and lipid plant components into amphiphilic molecules, that is, molecules that consist of separate hydrophobic (nonpolar) and hydrophilic (polar) parts or regions. The nonpolar parts of the molecules are composed of relatively unaltered segments of plant polymers and the polar parts of carboxylic acid

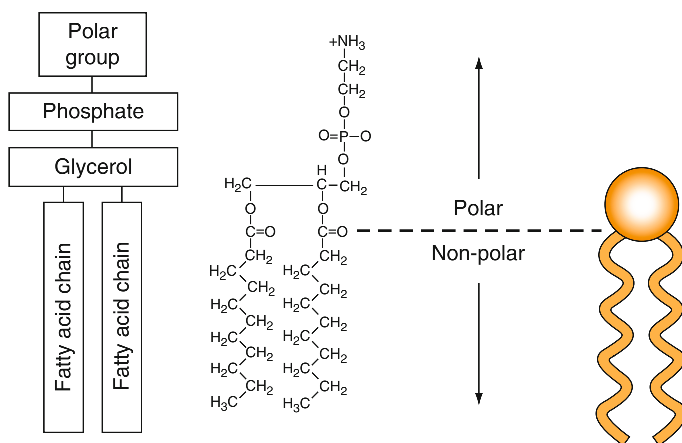


Figure 12 Example of amphiphilic molecule.

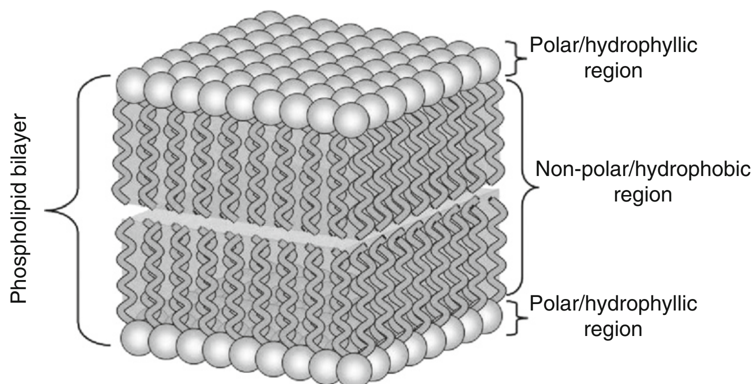


Figure 13 Example for a supramolecular assembly of amphiphilic molecules: Phospholipids assembled into membrane-like structures when immersed into a polar solvent.

groups. These amphiphiles form membrane-like aggregates on mineral surfaces and micelle-like aggregates in solution. The exterior surfaces of these aggregates are hydrophilic, and the interiors constitute separate hydrophobic liquid-like phases. By pushing the amphiphiles into aggregates, the soil solution achieves a higher state of entropy, because it can have more water molecules in a disordered state (compare Fig. 4).

4.6.2. Degradation or secondary syntheses: Far reaching implications

With his application of supramolecular chemistry to the organic materials in soils, Wershaw provided strong support for the degradative concept of organic matter transformations in soils, which had been dismissed by Kononova (1961) in favor of the competing synthetic concept. The *degradative* concept suggests that biological macromolecules are degraded to give rise to fragments which have structural and compositional features that can be related to their parent molecules. The *synthetic* concept assumes that humic macromolecules polymerize from small organic precursor molecules released from processes of metabolism of the soil flora and fauna, and from other breakdown products of natural macromolecules such as proteins, polysaccharides, and lignin (Hayes and Swift, 1990). As illustrated in Table 3, both concepts lead to fundamentally different molecular models for SOM.

All major functional features attributed to SOM on the basis of the assumption that it consists of humic substances that are polymeric, highly aromatic and recalcitrant lose their mechanistic basis, when a new structural model is applied. The differences between the two models can be even

Table 3 Brief description of competing models for the physical nature of SOM

The *humic polymer* model:	The *molecular aggregate* model:
The component molecules of SOM are produced by *secondary syntheses* reactions from degradation products. This assumption has led to models in which the NOM molecules are depicted as large, covalently bonded ("humic") polymers with unique chemical structures that are different from those of the starting materials.	SOM is composed of the partial *degradation* products of plant polymers. This "new view" has resulted in the development of models in which NOM is composed of (supra) molecular aggregates of the degradation products held together by entropic interactions and /or noncovalent bonds.
Molecular structure	*Molecular structure*
Fragments are joined by strong covalent bonds.	Amphiphilic fragments self-assemble through weak entropic interactions with water molecules.
Implication for stability	*Implication for stability*
The humic polymer model implies inherent resistance to decomposition, also called recalcitrance.	The molecular aggregate model does not provide a structural reason for inherent stability against decomposition.

better illustrated by comparing two graphical representations. Figure 14 shows the classical model of SOM as a mix of nonhumic biomolecules and macromolecular products of secondary syntheses. In the previous chapters we have presented several arguments in support of the position that the materials created in the course of alkaline treatments do not adequately represent the organic compounds which actually occur in soils. Figure 14 makes clear that the classical idea of SOM as consisting of "humic substances" loses its foundation as soon as this proposition is accepted.

The more recent Molecular Aggregate Model (Fig. 15) as proposed by Wershaw (2004) and supported by an increasing number of scientists (Burdon, 2001; Piccolo, 2001; Sposito, 2008; Sutton and Sposito, 2005), assumes that there are no products of secondary synthesis and that the majority of organic matter consists of relatively small organic fragments. This view interferes with important established beliefs within the science community. For example, classical definitions of humic substances mention their refractory nature or intrinsic resistance to decomposition. One of the explanations for inherent recalcitrance is the polymeric and macromolecular nature of humic molecules. If SOM fragments are not large polymeric macromolecules, then they either lose their alleged recalcitrance or this property needs to be explained otherwise. Thus a fundamental difference

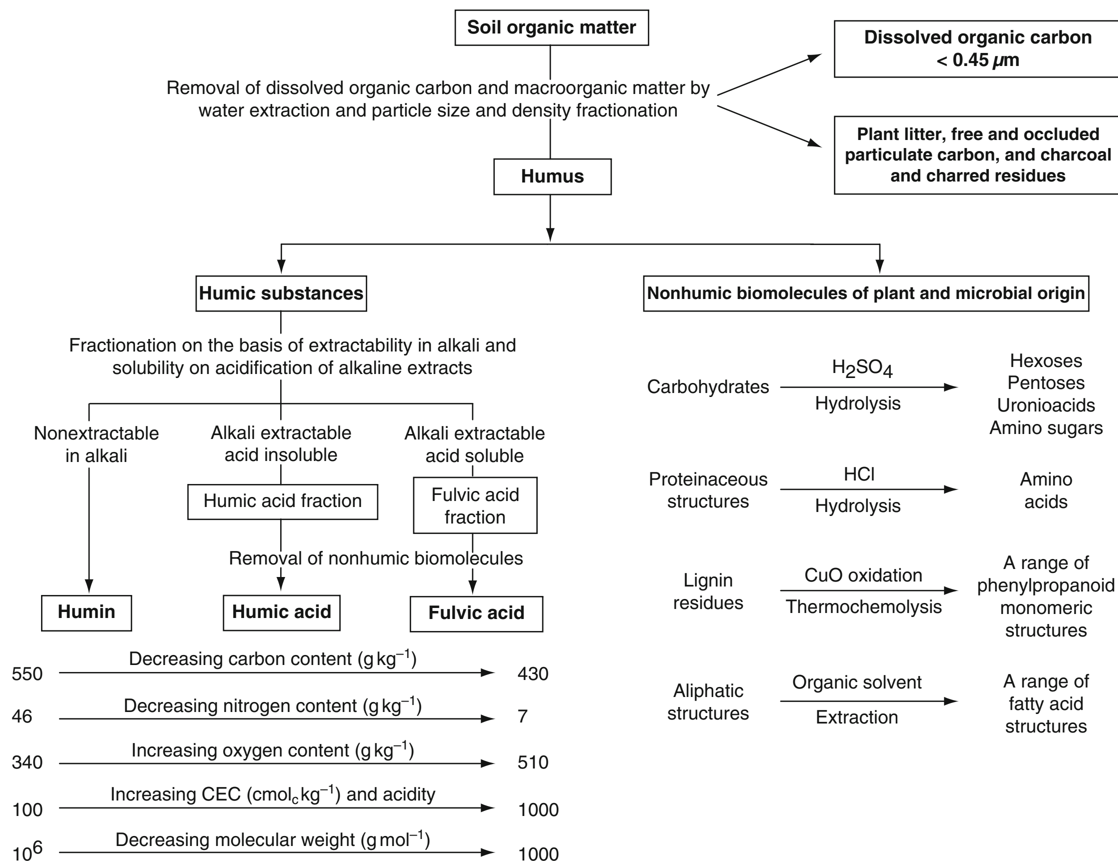


Figure 14 Soil organic matter according to the ''Humic Polymer'' model. Reproduced from Baldock and Nelson, 2000; Oades, 1989, with permission from Taylor and Francis.

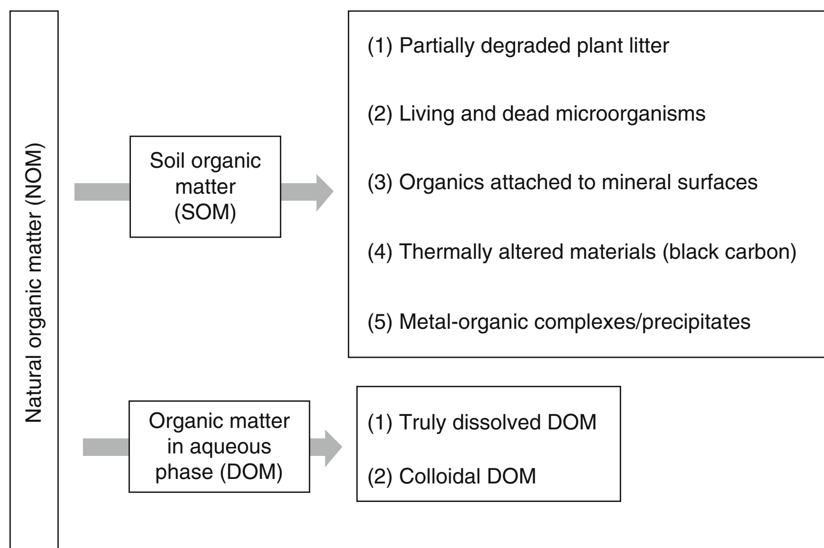


Figure 15 **Conceptual model of soil organic matter based on the "Molecular Aggregate" model (Wershaw, 2004): Note that the category "humic substances" is not used in this model.**

between the "Humic Polymer" concept and the "Molecular Aggregate" model is that the latter does not invoke the creation of refractory phases with extended turnover time. In fact, the experimental evidence gained in the recent past not only fails to identify distinct humic molecules even within alkaline preparations (Kelleher and Simpson, 2006), it also shows that humic substances (as obtained by alkaline extraction) decompose rapidly (Tatzber *et al.*, 2008).

Significantly, the Molecular Aggregate model was shown to be valid for compost leachates obtained without the use of alkali or any other extractant besides water (Wershaw *et al.*, 1995, 1996a,b).

From the standpoint of general plausibility, the "Molecular Aggregate" model is a much more convincing theory since it allows reduced carbon to move down a thermodynamic gradient towards CO_2 instead of requiring complicated and enigmatic syntheses pathways to create intermediate, recalcitrant macromolecular phases. Up to this point, no author has come forward to present a convincing thermodynamic explanation why a decomposer organism would invest the energy to synthesize a thermodynamically stable macromolecule as an intermediate product in the decomposition pathway.

The question of the correct molecular model for SOM issue is intensely (Burdon, 2001; MacCarthy, 2001; Piccolo, 2001; Swift, 1999), and sometimes fiercely (Haider, 1999, 2000; Schulten and Schnitzer, 2000) debated. Any future research involving the functions and the dynamics of SOM, be it

motivated by agricultural or more global biogeochemical questions, will have to make an assumption about the molecular nature of SOM. At present, this comes down to a decision between either the traditional Humic Polymer Model or the modern Molecular Aggregate Model. An extremely important implication of the Molecular Aggregate Model is that it would eliminate the complex and recalcitrant ''humic'' syntheses products with high activation energies that are at the center of current theories to explain the temperature dependence (Bosatta and Ågren, 1999; Davidson and Janssens, 2006) of SOM decomposition.

4.7. Glass transitions and the molecular structure of SOM

It has been posited (Schaumann, 2006a) that some arguments in the discussion regarding the ''correct'' molecular structure of SOM may have been derived from work with alkaline extracts (Conte and Piccolo, 1999) whose composition seemed to have been somewhat biased in favor of the Molecular Aggregate model. This view was combined with evidence of adsorption properties of humic substances (alkaline preparations) which could only be explained by the existence of some sort of polymeric matrix. When conducting adsorption experiments of apolar organic molecules to an alkaline preparation, LeBoeuf and Weber (1997) found that phenanthrene adsorption was best explained by simultaneous contributions of both adsorption and partitioning phenomena. The fact that their humic acid had a measurable glass transition point was taken as an indication that it contained glassy and rubbery domains, the former acting as a solid surface with finite adsorption sites and the latter acting as partitioning medium. In a follow-up publication LeBoeuf and Weber (2000a) further develop this analogy between SOM and synthetic polymers, suggesting that the organic matrices of soils and sediments should be divided into two categories manifesting mechanistically different adsorption behavior, that is, an amorphous, gel-like ''soft-carbon'' matrix or domain and a condensed, glasslike hard carbon matrix or domain.

A ''glass transition'' is a physical phenomenon associated with the transition of a polymeric material from a rigid or glassy state towards a more flexible, rubbery state as temperature increases. In order to exhibit a glass transition, a substance must be partially crystalline and partially amorphous in molecular structure. The glass transition temperature can then be seen as the temperature at which the degree of cross-linking within the polymeric material is substantially reduced. Observation of a glass transition in SOM thus means that the organic material contains polymeric structures.

While we agree with Schaumann (2006a) that neither polymeric macromolecules nor small molecules can and should be excluded as components of SOM, we do not believe that the observation of glass transitions is an argument in support of a secondary synthesis process which can create macromolecular ''humic'' substances. The presence of macromolecules

and polymers in SOM is well-documented (Burdon, 2001). Chilom and Rice (2005) showed that lipids can form crystallites, and that glass transitions observed for the materials under study were actually the melting of the crystallites present in the lipid fraction. Zhang *et al.* (2007) developed a diagenetic index to show that glass transition temperatures in organic materials are closely related to chemical characteristics of the organic samples as influenced by diagenetic or thermal alteration. Given the ubiquity of thermally altered materials (charcoal, soot) as well as plant derived biomacromolecules (crystalline lipids), we find that there are good reasons why SOM might exhibit glass transition behavior without requiring a "humification-type" process of secondary synthesis.

5. Reconciliation of Four Centuries of SOM Research

The amorphous, darkish appearance of SOM is similar in every soil on the globe. Humans do not have the sensory equipment to detect CO_2, the true end product of every aerobic decomposition process. Thus it has been customary to interpret SOM as the universal end product of decomposition. This belief has formed the foundation of the "educational initiation that prepares and licenses the student for professional practice" in the sense of Thomas Kuhn (1962) and over several centuries, science has tried hard to find mechanistic reasons why SOM should be stable in the environment. Early on, an operational extraction procedure was introduced (Achard, 1786) which created materials (humic substances) that were chemically highly undefined, but had some properties that were specific to certain taxonomic soil types (Kononova, 1961). The pioneers of soil radiocarbon found these materials to be depleted in ^{14}C (Campbell *et al.*, 1967) and concluded that this observation supported the paradigm that certain "humification" processes had created "stable" organic macromolecules out of "labile" plant litter precursor materials.

Mainstream science, as represented by the long list of textbooks that teach soil science (e. g. Brady and Weil, 2008), has chosen to adopt and promote the idea of SOM as a stable (recalcitrant, refractory) product of secondary syntheses, and this concept has found its way into soil carbon cycle models (e.g., RothC, CENTURY) where it influences how we project the effects of management and climate change on SOM. However, this traditional view of SOM neglects the fact that organic matter is a "thermodynamic anomaly atop a free energy precipice that drops off on all sides to dispersed, stable ingredients such as carbon dioxide, water, nitrate, and phosphate" (Hedges *et al.*, 2000) and is thus inherently unstable in the presence of a powerful electron acceptor like O_2. It is long known that

the true end product of decomposition is CO_2, and we have shown on the previous pages that the popular concept of SOM as an inert, biologically resistant material has not been substantiated. All available evidence shows that plant residues decay rather rapidly in aerobic soils with adequate moisture and temperature, and are more or less completely transformed, even the lignin fraction. Freshly incorporated litter carbon is comminuted through various processes and is either respired or enters the bodies of microbes (soil microbial biomass). The remnants of microbes and fungi, either as fragments of cell walls or as individual molecules, attach to mineral surfaces and can be protected against decomposition for centennial to millennial time scales. What is old is not necessarily old because it is stable, as presumed by the pioneers of radiocarbon in the early 1960s of the twentieth century. Sorptive protection (Mikutta *et al.*, 2007), aggregation (Salomé *et al.*, 2010), occlusion and facultative nonutilization (Ekschmitt *et al.*, 2008) allow seemingly labile organic materials to persist, rendering classical explanations for old age like enigmatic "humification pathways" obsolete.

5.1. What is SOM, what is it not?

1. Conceptually, SOM is a mixture of organic compounds in various stages of decomposition. There is not enough evidence to support the assumption that processes of secondary syntheses (humification) create recalcitrant polymeric macromolecules in relevant quantities.
2. Chemically and physically, the properties of SOM depend on the state factors as delineated by Jenny (1941) and thus vary between taxonomic soil types.
   - An important variable concerning the physical nature of SOM are anthropogenic additions, comprising materials as diverse as pesticides, municipal waste, charcoal, and soot.
   - Macromolecular and crystalline compounds do occur in SOM (Schaumann, 2006a), however, these either represent natural biomolecules and their decomposition products (Burdon, 2001) or have a diagenetic/pyrogenic (Zhang *et al.*, 2007) origin. Specific "humic" molecular structures have not been found to date (Kelleher *et al.* 2006; Lehmann *et al.* 2008).
   - The products of the alkaline extraction procedure (humic substances) are different from natural organic matter and should not be used as proxies for organic matter dynamics.
   - Molecular fragments are best portrayed as amphiphiles (Sutton and Sposito, 2005), with oxidative decomposition adding oxygen-containing functional groups to nonpolar molecular domains as decomposition progresses, thereby enhancing aqueous solubility.

5.2. How do the physical models of SOM affect assumptions about mechanisms?

5.2.1. Implications of amphiphilicity

Prior to the advent of high-resolution imaging techniques with spectroscopic capabilities, like scanning X-ray microscopy (STXM), mineral–organic interactions were typically conceptualized as two-dimensional, treating the mineral matrix as the sorbent and assuming humic polymers to represent the sorbate (Leinweber and Schulten, 1999a; Stevenson, 1985). Mechanistically, early models represented organo-mineral complexes as associations of large, multifunctional humic polymers with mineral surfaces via a broad range of bonding mechanisms (Fig. 16).

Over the last two decades, it has become apparent that such a model cannot account for a variety of dynamic features of mineral–organic complexes (Kleber *et al.*, 2007). Methods are now available that can directly look

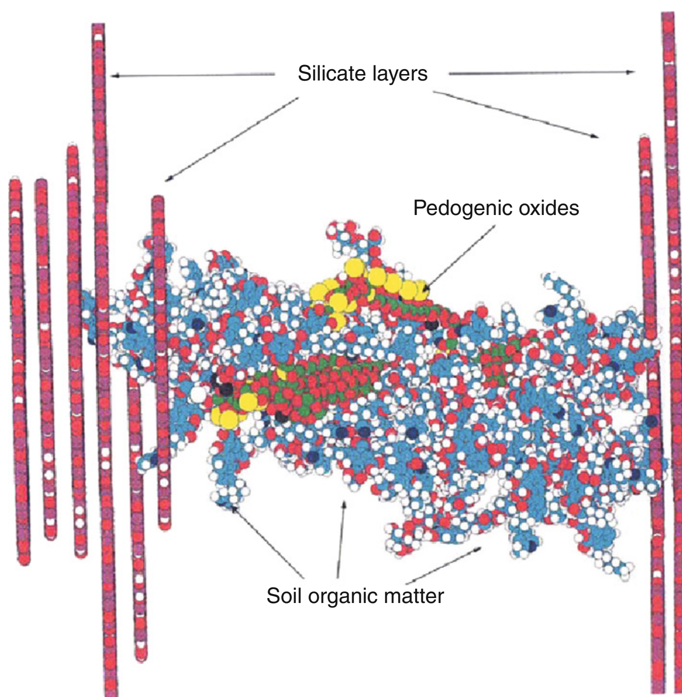


Figure 16 Model of a mineral–organic complex consisting of a mineral matrix of silicate layers and a bound soil organic matter complex of humic acids, with trapped labile carbohydrates, pedogenic Fe-oxides and alkali and earth alkali cations. Image corresponds to Fig. 9 in Leinweber and Schulten (1999a). Reprinted from Journal of Analytical and Applied Pyrolysis, 49 (1-2), P. Leinweber, H. -R. Schulten, Advances in analytical pyrolysis of soil organic matter, Pg :25.1999 Copyright (1999), with permission from Elsevier.

at the chemistry of SOM with nanometer resolution. When employing STXM coupled with near edge absorption fine structure spectroscopy (NEXAFS), Lehmann *et al.* (2008) were able to show that molecules with the same chemistry as those obtained in an alkaline extraction of soil were not detectable in whole soil. Rather, organic matter in soils was found to consist of identifiable and distinct molecular forms such as microbial and plant biopolymers not only in complex mixtures but in very complex spatial arrangements. The sum of these compounds yielded a signature similar to the alkaline extract, but individually they were very different from the molecular characteristics of the alkaline soil extract.

Two examples below illustrate how a conceptual model that assumes SOM to consist of a mix of mobile amphiphilic fragments rather than of polymeric macromolecules serves better to explain recent observational data.

5.2.1.1. Water repellency of soils Horne and McIntosh (2000) attributed seasonal changes in soil wettability to the existence of multiple, functionally different layers of organic matter on sandy grains. In their model, an inner-layer of more hydrophobic compounds is covered by a layer of amphiphilic compounds. When the soil is wettable, the hydrophobic material is effectively screened, especially if the outer surface is well hydrated. Water repellency occurs when the hydrophobic compounds making up the inner-layer are more exposed. This can be linked to seasonal moisture differences through a combination of factors: during the wet season, the outer layer will be well hydrated, amphiphilic compounds will be arranged in such a fashion as to have their polar portions oriented outwards, the carboxylate anion will predominate, and hydrophobic groups will be effectively screened by more hydrophilic groups. With the onset of the dry season, a more hydrophobic outer surface develops as the amphiphilic compounds change in orientation, the carboxylate groups are protonated, and contraction of the amphiphilic "surface screen" exposes the interior hydrophobic layer. In a recent contribution, Diehl *et al.* (2009) showed that the changes in the arrangement of organic molecules following variations in moisture content that were postulated by Horne and McIntosh (2000) are detectable. They concluded that changes in repellency result from an interplay between (i) water content driven changes, probably induced by changes in microscale moisture distribution and thickness of the water film, and (ii) changes in SOM surface chemistry, promoted by reorientation of amphiphilic moieties (Diehl *et al.*, 2009).

5.2.1.2. Variations in ^{14}C-based turnover times within one physical fraction Densiometric fluids provide a means for separating soils into fractions based upon particle density. Organic C in the dense fractions (e.g., >1.65 kg L^{-1}) of soils is commonly seen as being closely associated

with minerals and typically more depleted in ^{14}C than light-fraction material, which is not intimately bound to minerals (Trumbore and Zheng, 1996). However, Trumbore (1997) reports that the residue after repeated hydrolysis of mineral-associated organic matter in acids and bases still may contain some fast-cycling carbon. When Swanston *et al*. (2005) traced an atmospheric ^{14}C pulse into a forest ecosystem, they noticed that the elevated ^{14}C signal appeared not only in the light fraction, but also in the strongly mineral-associated dense fraction. They interpreted this observation as indicating the existence of at least two different C pools in the organo-mineral fraction: an older, more stable pool of C, and a younger, fast-cycling C pool. Such dynamics are easier to reconcile when SOM is assumed to encompass organic molecules that are mobile because of relatively small size and their amphiphilic nature, allowing them to be arranged in flexible supramolecular structures by interaction with the polar solvent water and other sorbed organic molecules.

5.2.2. Mineral: Organic interactions are most likely microbially mediated

Initiated largely by the work of Chenu and collaborators (Chenu and Plante, 2006; Chenu and Stotzky, 2002) and further supported by recent imaging and X-ray spectroscopic work (Lehmann *et al*., 2007, 2008; Wan *et al*., 2007), a functional view of carbon turnover dynamics at the micro-scale has gained traction over the last years that goes beyond the mineral sorbent—organic sorbate idea of previous years. In this view, the two-dimensional concept of soil as a system composed of adsorbent (mineral surface) and adsorbate (soil organic matter) has been extended to a three-dimensional view of soil consisting of a multitude of largely independent mineral–organic microstructures formed around colloidal to subcolloidal size organic matter nuclei, especially microbial/fungal cell fragments that are accompanied by some extracellular glue-type materials like polysaccharides and proteins. This view is supported by abundant evidence that substantial parts of mineral surfaces are not covered by organic matter (Arnarson and Keil, 2001; Kahle *et al*., 2002; Mayer and Xing, 2001; Ransom *et al*., 1998), a fact that is difficult to reconcile when the mineral matrix is conceptualized as an adsorbent and SOM as an adsorbate that is dissolved and/or suspended in the soil solution. Application of sequential density fractionation showed an increase of microbial signature in organic materials with increasing degree of mineral association (Sollins *et al*., 2006). Direct observation using state-of-the art NMR—techniques (Simpson *et al*., 2007) showed that SOM was predominately of microbial origin, either in the form of preserved material or intact cells. Carbon stabilized onto mineral fractions was assessed to have a distinct composition related more, to microbially processed, organic matter than to plant-related compounds (Grandy and Neff, 2008). All these reports are easier to reconcile with active processes of

microbial—mineral association than with the adsorption of a random mix of organic materials dissolved or suspended in the soil solution.

5.2.3. SOM turnover models and soil carbon stabilization

We have presented converging evidence (Bol *et al*., 2009; Marschner *et al*., 2008; Mikutta *et al*., 2006) that SOM stabilization is not well explained by the classical view that a humification process creates recalcitrant humic substances which then resist decomposition. Old SOM may contain significant proportions of chemically and thermodynamically labile carbon (Trumbore, 1997). This insight questions the general relevance of the stabilization mechanism of recalcitrance (Sollins *et al*., 1996) as a prerequisite for long turnover of SOM. However, predictive simulation models have historically treated the carbon pool that turns over slowly as the ''refractory'' carbon pool (Falloon *et al*., 2000). Predictive models such as CENTURY and RothC typically create several functional organic matter pools by assigning different decomposition rate constants to organic input materials based largely on the concept of recalcitrance (Jenkinson, 1990; Parton *et al*., 1987). The transfer of carbon from fast-cycling pools into pools with progressively longer turnover times is treated as a consequence of humification processes leading to the preservation of refractory substances (McGill, 1996; Parton *et al*., 1994). Thus, although clay content is recognized as another control on turnover time, models place strong emphasis on the assumption that slow and very slow turnover results from the recalcitrance of humified materials. However, potential feedback responses of the global soil system to climate warming will likely be much different if the carbon contained in the passive soil carbon pools is inherently resistant to decomposition or instead a ready substrate for decomposers stabilized only by its association with mineral surfaces. This illustrates why the physical and chemical nature of SOM is an important issue, and why it is unsatisfactory to use operational categories (humic substances) derived from arbitrary extraction procedures as proxies for the characterization of the organic materials in soils.

5.3. The bright side of alkaline extracts and humic substance preparations: Practical applications

While humic substances obtained through the procedure of alkaline extraction are poor proxies for the overall chemical structure of SOM, they do have a great potential to be developed into engineered natural organic sorbents for environmental applications with considerable commercial value. Mechanistically, carboxyl groups are formed during the enzymatic and oxidative depolymerization of plant biopolymers (Essington, 2003), increasing the solubility and extractability of organic materials as their decomposition progresses. The standard alkaline extraction procedure (using 0.1 M NaOH) ionizes all oxygen-containing functional groups in

organic substances that have p*K*a values below 13. This includes not only most alcoholic and phenolic –OH, but also virtually every carboxyl group. Thus the alkaline extraction procedure selects for highly decomposed materials with abundant oxygen-containing functional groups. The amphiphilic nature of these molecules allows them to interact with both polar and nonpolar counterparts, especially when the precursor materials are rich in aromatic moieties. Empirical studies have shown that alkaline preparations can be used in a surprising variety of practical applications, ranging from plant growth promoting soil amendments over soil remediation applications to sophisticated biomedical technology.

5.3.1. Alkaline extracts (humic substances) as soil amendments

Organic materials created by the alkaline extraction procedure can have beneficial effects on plant growth. A thorough review of respective laboratory and field experiments has recently been presented by Nardi *et al.* (2009). These authors distinguish three areas where an influence of alkaline preparations on plant growth has been shown. These are (i) morphological changes of plants (Nardi *et al.*, 2002), (ii) enhancement of nutrient use efficiency and (iii) effects of alkaline preparations (humic acids) on biochemical pathways like glycolysis (Nardi *et al.*, 2007). While beneficial effects have been demonstrated repeatedly, the indeterminate chemical nature of the alkaline preparations is seen as an obstacle to a more mechanistic understanding of causal relationships (Nardi *et al.*, 2009).

5.3.2. Alkaline extracts in soil remediation

5.3.2.1. Immobilization of organic contaminants and heavy metals If added to soil, alkaline preparations of decaying organic materials can adsorb and thus immobilize both organic and inorganic compounds, like polycyclic aromatic hydrocarbons and heavy metals. For example, Perminova *et al.* (2001) found an observed reduction in toxicity of polynuclear aromatic hydorcarbons (PAH) on *Daphnia magna* in the presence of alkali extracted humic substances. They were able to show that the detoxification efficiency correlated with the aromaticity of the alkaline preparations used. In an attempt to reduce the mobility of actinide elements in the environment, Shcherbina *et al.* (2007) used quinonoid enriched humic derivatives to complex and/or reduce Np(V) present in solution. These ‘‘designer humics’’ are essentially derived reducing agents that can serve as reactive components of a novel humic-based remediation technology. Similar concepts are evident in attempts to develop ‘‘engineered natural organic sorbents’’ for environmental applications (Tang *et al.*, 2008; Weber *et al.*, 2006). A recent review (Perminova and Hatfield, 2005) gives a comprehensive assessment of the concept of ‘‘designer humics’’ to remediate polluted environments.

5.3.2.2. Alkaline preparations as surfactants Alkaline soil extracts have surfactant-like properties owing to their amphiphilic chemical nature. This renders them capable of mobilizing certain contaminants. Conte *et al.* (2005) were able to show that an alkaline leonardite extract could be used in washings of a contaminated soil with the same efficiency as that of synthetic surfactants which exhibit some degree of biological toxicity. Similarly, alkaline extracts have also been shown to affect the dispersion of carbonaceous nanomaterials in aqueous environments. The presence of alkaline extracted materials greatly enhanced the dispersion of Fullerenes (C_{60}) and the dispersion process was further accelerated by sunlight (Li *et al.*, 2009). Chappell *et al.* (2009) showed that carbon nanotubes were stabilized in suspension to a varying extent by the addition of different types of alkaline extracts.

5.3.3. Biomedical applications of alkaline preparations

Alkaline extracts in various forms have been shown to possess antibacterial, antiulcerogenic, antiallergic, and antiinflammatory properties (Schepetkin *et al.*, 2002). Recent *in vitro* experiments have demonstrated that certain kinds of alkaline extracts (e.g., potassium humate) may inhibit the production of inflammatory Cytokines. Films made from alkaline extracts and grown via layer-by-layer self-assembly were presented as a plausible membrane material for implantable glucose sensors (Galeska *et al.*, 2001). Humic acid based assemblies were found to be soft and pliable, qualities that are important to minimize tissue damage. The films were considered biocompatible since *in vivo* studies in rats indicated only mild tissue reactions.

5.3.4. Alkaline extracts have a future in technology, not in biogeochemistry

We conclude with the observation that even the small selection of technological applications for alkaline extracts presented here demonstrates the potential for the development of engineered natural organic materials out of SOM extracts. One prerequisite to success will be the continuation of efforts to increase knowledge about the chemical nature of alkaline extracts as influenced by the extraction procedure and an improved mechanistic understanding of their interactions within environmental systems. However, we do not recommend using alkaline extractions (a.k.a. humic substances) as proxies for natural SOM in future studies of soil biogeochemistry.

6. Outlook for the Future

As scientists interested in coming to a more complete understanding of SOM and the factors that control its stabilization and destabilization we recognize that we stand on the shoulders of many other scientists who have

conducted research on SOM over the past two centuries. Our purpose in writing this paper was to consider their numerous contributions in the light of more recent developments. It was never our intention to impugn the work of anyone, but to build upon their work and to try to frame it in such a way as to constructively highlight new and emerging ideas about what SOM is and what it is not.

6.1. What are implications for future research?

While we have a good general picture of what SOM is, much remains to be learned about it, the processes that alter and change it, and the factors that affect its stability. It is exciting, however, to consider the advancements that are currently taking place in SOM research. With every publication new information is gained that either confirms previous theories or leads to the development of new ones. With the currently available technologies an increasing number of papers are being published on the nature and chemistry of SOM and the factors that affect its stability and turnover. The use of multiple concurrent measurement techniques provides the means for ''seeing'' many features of SOM simultaneously (Lehmann *et al.*, 2008). Some techniques even allow measurements to be made on a whole soil sample, which is making measurements *in situ*, and might be the ideal way to view soil processes. Without the ability to see too far into the future, it is hard to know precisely where research on SOM will go in the next 10–20 years. It is exciting to contemplate the advancements that will take place as scientists continue their research and new young scientists begin their own investigations.

It is important, however, to not overlook older methods and the kinds of data that they provide. For example, infrared spectroscopy has been around for a long time. Coupling newer FT-IR (Fourier transformed infrared) instruments with computers that have spectral interpretation and quantification software provides a powerful tool for ''seeing'' the constituents of SOM and the effects of various treatments on them. Simple column leaching studies can provide a lot of information about the sorption and movement of organics through soils. Stable isotopes (e.g., ^{13}C ^{15}N ^{18}O) and radioactive isotopes (i.e., ^{14}C) can provide insights about the cycling, processing, and residence time of SOM.

Besides funding and gaining access to sophisticated instruments there are few limitations on the future of SOM research. It will depend on the creativity of the scientists working in this area and their awareness of past and present research. It will also depend on their ability to assess the research needed to solve real-world problems. As will be discussed in the following sections, there are important environmental and social issues that will need to be solved by understanding SOM and how to manage it and how to sequester it. Soil scientists, agronomists, soil chemists and environmental

scientists need to partner and to define the problems and gather the data needed to solve them. We believe that the solution to a number of these issues will be found in a more complete understanding of SOM.

6.2. What are potential environmental implications?

One of the biggest current environmental issues is that of global climate change. The causes are thought to be anthropogenic, but natural variation cannot be overlooked (IPCC, 2007). Whatever the cause, the central theme is that increased atmospheric concentrations of greenhouse gasses are causing global surface temperatures to rise (IPCC, 2007). Carbon dioxide is a key greenhouse gas that is produced by combusting fossil fuels and by the decomposition of plant material and oxidation of SOM. On an annual basis, the fixation of atmospheric CO_2 by plants and the production of CO_2 through plant respiration and oxidation of SOM are roughly in balance (Schlesinger and Andrews, 2000). However, this has been changing as soils are being kept in continuous agriculture and forests are being cleared for urban development and agriculture.

Soils are the largest terrestrial nonfossil reservoir of carbon. It is estimated that world soils hold approximately 1500 Pg of organic carbon within the first 100 cm (Schlesinger and Andrews, 2000). The atmosphere holds 750 Pg of carbon as CO_2. Global vegetation holds approximately 560 Pg of organic carbon. A key part of the climate change question is what will be the effect of a warmer climate regimen on the vast stock of soil carbon? Will it lead to increased oxidation of SOM? Will this cause a positive feedback with the atmosphere, that is, warmer atmosphere leads to great soil carbon oxidation that leads to a warmer atmosphere? Understanding the factors that stabilize SOM may lead to better soil management strategies that help maintain current levels of SOM or ways to increase the amount of carbon held in soils.

A related topic is the ability to use computer simulation models to predict the future effects of environmental stressors such as climate change. Numerous models of carbon cycling, both above- and belowground, depend upon an accurate understanding of the carbon cycle. While there are no perfect models, they rely heavily upon the knowledge of SOM that soil scientists and chemists provide. An incorrect "model" of SOM will lead to an incorrect simulation of what will happen to SOM. The implications of this could be large. It is imperative that we as a discipline participate in this work and contribute to developing an accurate picture of how SOM accumulates and is lost.

Another area of fertile research is quantifying the services that SOM provides to people. While a little odd at the outset, ecosystem services are "the output of ecosystem functions or processes that support (directly or indirectly) human welfare or have the potential to do so in the future"

(EPA, 2006). Following this definition, SOM provides ecosystem services. For example, it is well known that soils high in organic matter are less likely to erode (wind or water) than soils with low organic matter contents. This is a "service" that SOM provides. SOM imparts other services such as cation exchange capacity, nutrients, improves soil tilth, water infiltration and storage. It also helps in pesticide sorption and degradation, and has a role in waste isolation and degradation, as well as a role in water cleansing. The ability to quantify the ecosystem services imparted by SOM will likely have an important role in future environmental protection and where lands are to be managed and developed. This will rely upon an understanding of SOM and the factors that control its accumulation or loss.

6.3. Are there implications for agriculture and sustainability?

SOM has a number of distinct roles in agriculture as we have described above. It is important for soil tilth, nutrient supply and cycling, water infiltration and supply and biodiversity. At the same time it also has a role in sustainability of natural systems. Sustainability is a term that is a lot like the definition of ecosystem services, but goes further in that it depicts the ability of a system to maintain the supply of services. For example, how long can a soil be used in continuous agriculture? The answer depends upon a lot of things, but it is well known that maintaining the status of SOM will keep this soil in production longer than if the soil were to become depleted in organic matter. The same is true for forest production and all of the other services that soils provide.

With a burgeoning world population and a dwindling supply of fossil fuels there is increasing pressure on soils to provide more food, fiber, and clean drinking water than ever before (Lal, 2009). It is becoming increasingly important that soils be managed in such a way as to maintain the stocks of SOM. This means that organic residues need to be incorporated into soils. Crops should be grown in rotations that include legumes. Not only does good SOM management help sustain agriculture, it will also help keep carbon in the soil and may lead to additional sequestration that reduces the amount of greenhouse gasses in the atmosphere.

6.4. How should the "New View" of the physical nature of SOM be presented in educational settings and educational literature?

It is important that students being trained in soil science, agronomy, ecology, environmental science and civil engineering have an accurate picture of SOM. As people who will depend upon their knowledge of soils and SOM as they make decisions and design remediation systems, or in other applications, knowing the best possible information will be to their benefit.

It is likely to also benefit those who will depend upon the decisions and judgments of the professional scientist or engineer. The information needed by someone who is taking an introductory course in soils for the purpose of becoming a land manager may be different from someone being trained to be a soil chemist, yet both need current information. It is time that introductory texts begin to incorporate current information on SOM if these texts are to continue to be used. Once outdated information is imprinted in a student's mind, it is difficult to replace it with newer information, especially if the newer information is more complex. It is a disservice to students to not provide them with current information as it leaves them at a disadvantage since they will need to use the best available information when they rely upon it for making management or environmental decisions.

With the continuing controversy about the chemical and physical nature of SOM, how should this information be presented to the beginner? We recommend that a historical perspective on SOM be included, similar to what we have presented here. From there, additional information should be provided that builds upon the early work and brings students up to the modern concepts and understanding of SOM. We realize that the degree to which this information is provided should be tempered by the audience since undergraduate students and graduate students will have different levels of need. This is certainly not a simple topic to cover, but we want to challenge and encourage instructors, professors, and textbook authors to provide as complete a picture as possible of SOM so that the next generation of soil practitioners is well prepared to make good applied decisions or to take the research on SOM to the next level.

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CHAPTER FOUR

Implications of the Knowledge Paradox for Soil Science*

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Contents

Abstract

Research results that could potentially provide a major contribution to innovation and sustainable development are often not accepted by or implemented in society. This, in short, is the knowledge paradox that is also relevant for soil science. The need to operate in a more interactive mode towards stakeholders

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and policy makers has been recognized in literature, has become more urgent in our Knowledge Democracy and is accepted by funding agencies. But true transdisciplinarity is difficult to realize and recommendations are discussed for soil science research to improve its effectivity by (i) focusing on unique niches, defined here in terms of seven basic soil functions, some of which have been preempted by others; (ii) considering the policy cycle as a point of reference; (iii) focusing on deriving a series of options (each one with economic, social and environmental tradeoffs) rather than on single solutions of problems related to sustainable development; and (iv) taking the entire knowledge chain into consideration from the tacit to the cutting-edge. Additional attention is suggested for communication and public relations in terms of (i) defining soil quality; (ii) adopting the ecological footprint concept and (iii) presenting storylines for major soil types. Transdisciplinarity requires working in "Communities of Practice" (CoPs); but first, the soil science profession should improve internal cohesion and cooperation by combating current atomization of subdisciplines and by defining mutual responsibilities in soil science "Communities of Scientific Practice" (CSPs) with special attention to quality control, education, and basic research, which is vital for the future of the soil science profession.

1. Introduction

Research results that could potentially provide a major contribution to innovation and sustainable development are all too often not accepted by or implemented in society. This, in short, is the knowledge paradox. There may be many reasons why this occurs. Existing rules and regulations may act as a brake on new developments as does a conservative, risk-averse attitude of a wide variety of stakeholders and policymakers. The different opinions and interests of all these participants, expressed in complex sustainable development debates, may easily lead to paralysis. Also, scientists should acknowledge that the way they have traditionally worked and communicated may not be in tune any more with recipients in today's society engaged with Google, Facebook, Twitter, Hyves, and other modern sources of information and means of communication. The knowledge gap is a reality for soil science. Some arbitrary examples could be cited: (i) Carbon sequestration in soils by adapting soil management practices has a higher potential than has been so far recognized in the overall climate debate; (ii) Regional land use debates are dominated by economic and political factors while important soil potentials play a minor role, if any, in the decision process; (iii) Precision agriculture is increasingly dominated by a technology-push, consisting of remote sensing and GPS-guided equipment, while processes in the soil system that determine water availability for crops and possible leaching of chemicals to the environment are increasingly ignored, even though they are crucial for fine-tuning soil management;

(iv) Soils provide ecosystem services and play a key role in environmental quality but they do not receive the same attention or funding as water and air. (v) Various international reports on food security mention soils in a generic manner, ignoring the widely different potentials of different soils in different climate zones; (vi) Combating environmental pollution often involves technical end-of-pipe solutions while research has shown that modified land management may produce important results upstream using the soil effectively as a living filter.

In the 1990s, the "Knowledge Economy" was seen as a glorious guide to the future, but the Knowledge Paradox has dimmed these prospects somewhat and recently, attention is shifting to: "Knowledge Democracy" (www.knowledgedemocracy.nl), where sizable groups of citizens have strong opinions on things, which they are quite willing to share 24 h a day. This has been made possible by rapid developments during the last decade of information technology via mobile phones and the internet, resulting in many new forms of possible communication and interaction.

Where does this leave science, and in this particular case, soil science? The new knowledge democracy probably requires new approaches to research and its communication. It is certainly not "business-as-usual" anymore. But what are we to do?

Funding agencies and scientific organizations are picking up the societal signals and are increasingly advocating and mandating intensive interaction with stakeholders and policymakers before and during research. For example, the International Council for Science (ICSU) states in its strategic plan for 2006–2010 (ICSU, 2005): ..."where science is used for the benefit of all... and where scientific knowledge is effectively linked to policy making." Their goal is to "strengthen the international science for the benefit of society." The implicit suggestion here is that benefits to society are realized when scientific knowledge is effectively linked to policy making. This sounds convincing but the linking process has so far not been well defined.

The research programs of the European Union (EU) and many other funding agencies require interaction with stakeholders and policy makers before and during the research process. But in contrast to traditional research with well established protocols, there is little guidance as to how to perform interactive forms of research while maintaining high scientific standards. Researchers duly include emphasis on interaction in their research proposals as they have little choice; however, they find it difficult to handle this requirement in practice. The recent report on Frontiers in Soil Science Research (NRC, 2009) recommends the development of new models for interdisciplinary collaboration and the translation of soil science research into information for stakeholders and end users but does not provide guidelines or examples. This chapter represents an attempt to make a contribution to this discussion by analyzing the possible implications

of the Knowledge Paradox, as indicated, for the soil science profession and to explore ways in which the observed trends can possibly be redirected. This may be particularly relevant at a time when research funding is decreasing, research groups dwindling, and researchers retreating to their own personal corners in the research arena which, they believe, might be the best way to survive. This is unlikely to be wise as it will probably result in even less interaction with outsiders and lower visibility.

2. Mode-1 Versus Mode-2 Research

During the last decade many papers and books have appeared covering the changing relationships between science and society. The influential book by Gibbons *et al.* (1994) distinguished traditional monodisciplinary mode-1 science versus transdisciplinary mode-2 science in which scientists of different disciplines work together with various stakeholders and policy makers. Mode-1 occurs in an academic context, while mode-2 operates in a context of application and is thus more problem oriented. Mode-1 is characterized by autonomy, based on the independence of science, while mode-2 is subject to social accountability. Also, quality control is different. Mode-1 research has traditional quality control in terms of the number of publications as expressed in citations and h-factors, while mode-2 research is judged by novel procedures also considering societal effectivity. The mode-2 approach certainly reflects societal concerns about the role of science, but many scientists are rightly concerned that an unqualified shift to mode-2 could involve loss of independence and scientific quality, which are both essential ingredients of science. An effort will be made in this chapter to address those particular, valid concerns.

To avoid confusion about terminology, the term interdisciplinarity is here used to describe the interaction between different scientific disciplines without stakeholder involvement. Hessels and van Lente (2008) pointed out that, using different terminology, many other authors have also indicated the need for increasing interaction between science and society. Bohme *et al.* (1983) speak about Finalization Science; Irvine and Martin (1984) about Strategic Research; and Functowicz and Ravetz (1993) about post-normal science. Edquist (1997) defines Innovation Systems, while Slaughter and Leslie (1997) describe Academic Capitalism. Ziman (2000) speaks about Post Academic Science and Etzkowitz and Leydesdorff (2000) about the Triple Helix. Ten years later, we must conclude that many academic studies have appeared covering interaction of science with society. But much less has happened in the real world where strategic plans and project requirements advocate the mode-2 approach, while many

scientists stick to their old habit of doing mode-1 research focusing on writing papers, which is considerably less time consuming than getting involved with a wide array of stakeholders and policy makers. There is, unfortunately, a painful gap between words and deeds. But as funding agencies are increasingly embracing the mode-2 approach, the scientific community would be well advised to reconsider its procedures so as to be better prepared for the future.

3. How to Define "Niches" for Soil Science When Studying the Big Environmental Issues of Today

Research on sustainable development implies consideration of not only environmental but also of economic and social aspects when studying a given problem (the well known 3Ps). This creates complicated conditions that are difficult to handle for researchers not only because highly diverse groups of researchers are involved, each one with their own research culture, but also because this already diverse group will have to communicate somehow with a quite diverse group of stakeholders and policy makers with quite diverse interests many of them of a commercial nature. So, how to realize in practice a desirable mode-2 approach that sounds so attractive on paper? A full discussion of this important topic is beyond the scope of this chapter. However, one element can be stressed here and that is the need for a clear identity of each of the disciplines being involved in terms of what contribution they can make to the overall discussion in the interdisciplinary groups studying sustainable development. The key question is: What are the core values (the "niches") that any given profession can contribute to the discussion on sustainable development? If such core values are not clear, the impact of that particular discipline in the overall discussions is likely to be minor and they may be sidelined.

What are the big societal issues for the next decade associated with sustainable development and what are the unique contributions to be made in each of these areas by soil science? A not necessarily complete list of societal issues follows: (i) energy; (ii) food security; (iii) climate; (iv) water; (v) environment, and (vi) biodiversity.

Discussions trying to define "niches" for soil science research occurred in the context of establishing the EU Soil Protection Strategy, later adopted in September 2006 (CEC, 2006). The overall objective of this strategy is: "protection and sustainable use of soil by preventing further soil degradation by preserving its functions and by restoring degraded soil to a level of functionality consistent with current and intended use." Some 60 soil

scientists from countries within the EU initiated a 4-year study to define research needs for this Strategy (ECSSS, 2004). Their recommendations focused on seven threats to soils: (i) erosion; (ii) local and diffuse contamination; (iii) loss of organic matter; (iv) loss of biodiversity; (v) compaction and other physical soil degradation; (vi) stalinization (floods and landslides); and (vii) sealing.

Research needs for each of these threats were defined for five clusters, applying the DPSIR system defining D(rivers) of land use change, the resulting P(ressures), the S(tate) of a given soil or landscape, I (mpact) of a given pressure, and R (espons) of a system being subjected to pressures:

Cluster 1. Processes underlying soil functions and qualities
Cluster 2. Spatial and Temporal changes of soil processes and parameters (State: "S")
Cluster 3. Ecological, economical, and social drivers of soil threats (Drivers: "D" and Pressures: "P")
Cluster 4. Factors (threats) influencing soil eco-services (Impacts: "I")
Cluster 5. Strategies and operational procedures for soil protection (Responses: "R")

This resulted overall in some 110 "Priority Research Areas" for soil protection and the management of Europe's natural resources (ECSSS, 2004). Though conceptually sound and containing many relevant and valid recommendations, the form of the analysis presented was not too convincing to policy makers and stakeholders as it appeared to avoid making choices while catering to a general suspicion that scientists are primarily interested to feather their own nests.

Aside from the fact that the term: "soil threat" is unfortunate (soils do not threaten anyone but are threatened themselves), it also has a negative connotation which is not attractive. Rather than react to negative signals by trying to reverse developments and point them into a positive direction, it is preferable to formulate soil potentials in a positive way in terms of what might be possible, defining certain well defined actions to be undertaken. In this context, soil functions are suitable to form the starting point for such an approach. Their character is neutral rather than negative. But more important is the fact that "threats" are being studied by many disciplines while also many groups of stakeholders are directly involved. "Threats" call therefore for transdisciplinary mode-2 research. Soil scientists will somehow have to find their "niche" in all of this when directly focusing on "threats." In contrast, soil scientists can first focus on studying soil functions and take expertise thus obtained next to the transdisciplinary groups studying: "Threats." This will allow them to better and more effectively define their particular contribution.

4. Studying the Seven Soil Functions

4.1. Introduction

The Royal Academy of Sciences of the Netherlands convened a study group exploring use of earth sciences for spatial planning (KNAW, 2008; see also Bouma *et al.*, 2008). One important difference with the ECSSS (2004) analysis was a primary emphasis on soil functions rather than on soil threats, both defined by CEC (2006). Seven functions are distinguished:

1. Production of food and other biomass
2. Storing, filtering, and transformation of compounds
3. Providing habitat and gene pool
4. Providing the physical and cultural environment for humankind
5. Source of raw materials
6. Acting as a carbon pool
7. Archive of geological and archeological heritage

Soil functions, together reflecting the 3Ps of sustainable development, provide ecosystem services that are valuable for society. Estimates indicate that they have a value for society of at least 30 trillion dollars (see NRC, 2009). Be that as it may, we find that some functions (nos 1, 5, and perhaps 6) are marketable, but others (nos 2, 3, 4, and 7) are not. The latter creates particular problems when financing research and requires substantial efforts to generate broad support.

Different soil functions can contribute to studies on the various societal issues, mentioned in Section 3 above, as follows: Functions 1 and 5 are particularly relevant for the energy issue. Functions 1 and 6 for food security; function 6 for climate issues; functions 2, 4, and 7 for environmental issues; function 3 for biodiversity and functions 1, 2, and 4 for the water issue. Clearly, different soil functions have a different relevance for societal issues related to sustainable development, but they all have a clear function thereby providing specific "niches" for soil research focused on sustainable development.

Soil scientists should realize that other researchers also focus their efforts on the societal sustainability issues and they compete in obtaining available research funds: Genetically modified crops are being promoted to increase biomass yields for biofuels also in very dry or salty soils. Soil conditions are usually not considered by genetic engineers but only integrated approaches of the entire production system are likely to produce sustainable results. Also, as long as potential crop yields are much lower than actual yields much is still to be gained by improving management of existing cropping systems. This is also relevant for food security, where a second factor plays a role as well. Economists point out that food security is not so much a problem

of production but more of distribution and buying power. There is enough food but some people cannot afford to buy it. This is correct, as such, but the expected 9 billion world inhabitants in 2040 require a doubling of current food production and this is certainly not a matter of only distribution and buying power. One option to mitigate the effects of climate change is to sequestrate CO_2 in geologic formations. Billions of dollars and euros are being spent to explore this option. But this is not only very costly but the associated risks are as yet unclear and sequestration of carbon in suitable soils by adapted soil management could be a more effective procedure, the more so since higher soil organic matter contents increase soil quality (SQ): a double benefit. When combating environmental issues, there is considerable emphasis on technical end-of-pipe solutions. Many of them are useful but much is to be gained by adopting practices of precision agriculture that fine-tune additions of agrochemicals to crop needs as a function of time, thereby minimizing losses to the environment and the associated pollution. This is also helpful in promoting biodiversity by avoiding local over-fertilization with adverse effects on the natural vegetation. Finally, studies of water issues can be improved by including better soil processes in comprehensive models because soil can be quite effective in purifying percolating water under proper management conditions, before the water is released into ground or surface water. Some hydrological modeling studies are less effective than they could be as they do not include an up-to-date soil module. Knowing competitors and their intentions, as broadly outlined above, is necessary as it allows development of more effective marketing procedures demonstrating particular advantages of, in this case, soil science-based approaches.

4.2. Research needs for the soil functions

4.2.1. Function 1: Production of food and other biomass

Simulation models are now widely used to predict crop yields as a function of crop characteristics and actual and potential environmental conditions. Two elements are particularly significant for soil science: (i) the contact between roots and soil, and (ii) water in the root-zone in space and time. Many simulation models, like the widely used DSSAT family (Jones *et al*., 2003), have simple expressions for soil. They use the ‘‘tipping-bucket’’ concept assuming water to be completely available only between two pressure heads: field capacity (often 0.3 bar) and wilting point (15 bar) (h2, and h4, respectively, in Fig. 1). The evaporative demand by the plants can only be completely satisfied within this pressure range. When it rains or during irrigation, the root-zone will fill up with water until field capacity. When more water infiltrates, it will discharge instantly to the subsoil and ‘‘disappear.’’ Contact between soil and roots is assumed to be perfect, while the root-zone has a given thickness. Feddes *et al*. (1976) introduced a more realistic ‘‘sink-term,’’ later applied in their SWAP model (Kroes *et al*., 2008)

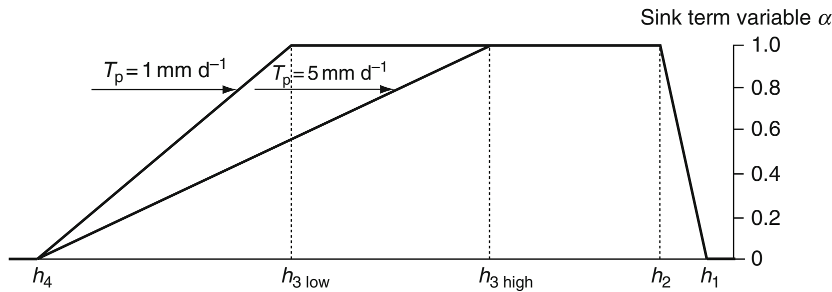


Figure 1 The sink term variable alpha (Feddes *et al*., 1976), which defines the reduction of potential transpiration as a function of the pressure head (h) in the soil and of the transpiration rate T_p.



Figure 2 Roots concentrated on ped faces, which may lead to unaccessibility of otherwise available water inside the peds in terms of its pressure potential.

assuming maximum availability of water only between two pressure heads (h2 and h3) that vary by crop and by evaporative demand, while decreasing gradually beyond these two points until h4 is reached and h1 on the "wet" end of the scale (Fig. 1). Thus the rigid and abrupt h2–h4 concept of "available water" is significantly refined, yielding much better results. Two additional problems may, however, occur also when using this refined model: (i) The implicit assumption that contact between roots and soil is "perfect" in the root-zone is often not realistic. Soil surveyors have often observed concentration of roots on faces of structural elements ("peds") (Fig. 2). This implies that water at a given depth in the soil may be "available" according to the standard model but not "accessible." Droogers *et al.* (1997) showed this effect to be significant and presented a method to characterize the "accessibility" factor. Field observations confirm that sometimes plants wilt in structured soils, particularly at relatively high

evaporation rates, while according to standard simulation models there should be plenty of "available" water. (ii) water infiltration and redistribution, as calculated in standard physical simulation models, is assumed to occur in a homogeneous and isotropic manner. In other words, the water content is the same at each depth at any given time, and water extraction occurs in the entire root-zone. This may more or less be correct in relatively homogeneous sands but not in structured soil. Even in sands, hydrophobicity results in heterogeneous infiltration patterns (e.g., Ritsema *et al.*, 2005) with major effects on water uptake by roots. Bouma and Dekker (1978) have demonstrated infiltration patterns in dry clay soil, consisting of micro-relief-governed infiltration at the soil surface into vertical, continuous cracks between peds, forming small bands of free water flowing vertically downwards along ped faces. Depths of one meter or more were reached in just a few minutes and water may collect in dead-end pores at different depth in the soil. The same effect was reported by Bouma *et al.* (1982) for silty soils with worm channels. The process of free water moving rapidly along macropores through an unsaturated soil matrix is called "bypass flow." A simple field measurement method has been developed (Booltink and Bouma, 2002). A few deep roots may follow deep percolating water that accumulates in pockets deep into the soil and this mechanism explains the common field observation that crops are not wilted on soils where water in the root-zone in the upper dry part of the soil is not available anymore. Clearly, current simulation models do not consider these important field mechanisms and input from pedologists could help modelers to improve their representation of water uptake by roots under natural conditions in the field.

There is a tendency in crop growth models to focus on plant aspects: evaporation at different growth stages, morphology of leaves etc. This is to be expected from crop physiologists but the rest of the model should have a corresponding degree of detail for the entire model to be well balanced. If soil input only consists of estimates of available water and depth of rooting, data can easily be supplied by non soil scientists using available databases and pedotransferfunctions relating soil texture, bulk density and organic matter content to water availability (Pachepsky and Rawls, 2004). Soil scientists are not needed for such activities and in order to prove the relevance of their soil expertise, soil scientists should demonstrate that such simple models of water uptake provide incorrect and misleading results for many soils in the field and that more detailed research, as indicated above, is needed to allow better modeling. So far, soil scientists are far too relaxed about this and do not realize that easily accessible databases make their specialized expertise redundant if simple approaches continue to be used by non soil scientists. This will happen if soil scientists do not challenge such simplified approaches!

4.2.2. Function 2: Storing, filtering and transformation of compounds

Just as with function 1, where use of simple expressions and easily available data could make specialized input by soil scientists irrelevant, function 2 can also simply be expressed by the Cation Exchange Capacity, the Organic Matter content and, perhaps, the air content at field capacity or some other simple measure for aeration. Such values can rapidly be derived from databases by any interested individual and comparison with concentrations of chemicals in percolating liquid can yield easy estimates of adsorption assuming complete contact. Adequate characterization of function 2 needs, however, a more sophisticated approach to be provided by soil scientists. Two aspects will be discussed: (i) the importance of travel times as an indicator for transformation of compounds, and (ii) the importance and implications of soil heterogeneity.

The importance of travel times is illustrated in Fig. 3. A solution with artificially high concentrations of poliovirus was percolated through a sand column at two flow rates. At 50 cm/day, a substantial quantity of virus left the column. At 5 cm/day, however, viruses were removed within 50 cm, as was observed when sampling the column after termination of the experiment. No virus particle ever left the column. Slower travel times imply better and longer contact between the solution and the soil particles because the liquid moves through only the smaller pores as larger pores are filled with air in unsaturated soil. Each soil material has a characteristic relationship between travel times and purification and this information has been used to design innovative soil disposal systems of septic tank effluent (e.g., Bouma, 1979). Structured soils offer special problems. Water will generally

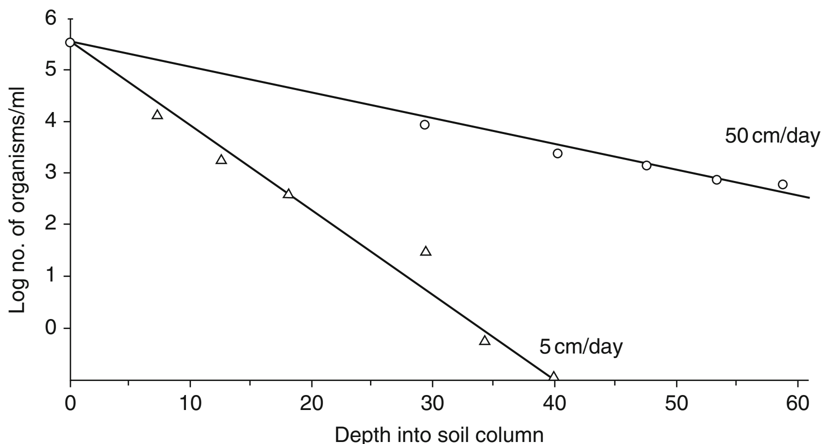


Figure 3 Purification of septic tank effluent spiked with high concentrations of poliovirus at two flow rates in a 60-cm long sand column (Bouma, 1979).

pass slowly through and more rapidly around the structural elements (peds), while it only moves through the peds below a characteristic flow rate for each structure type. This has implications for purification as was demonstrated for a well structured silt loam soil (Fig. 4). At flow rates of 5 mm/day, fecal coliforms passed through a 50-cm long column, while they did not at 3 mm/day when the solution only moved through the peds and more rapid flow in relatively large pores around the peds could not occur, leading to longer contact times and better purification. Each soil structure type has a typical threshold flux, separating flow through, rather than around the matrix. This has, as yet, not been explored and large databases with soil structure descriptions are ready to be analyzed. Simulation models for chemical transport in soil are based on the implicit assumption of physical flow models that show the soil is isotropic and homogeneous. Some soils are more anisotropic than others but none of them are truly homogeneous and isotropic. Young in NRC (2009) used modern scanning and tracing techniques to show that often <0.01% of the soil volume is occupied by so called biofilms that are active in chemical and microbiological transformations occurring in "hot-spots." Bouma *et al.* (1979) showed, using methylene blue dye to trace vertical saturated flow patterns in a clay soil, that the

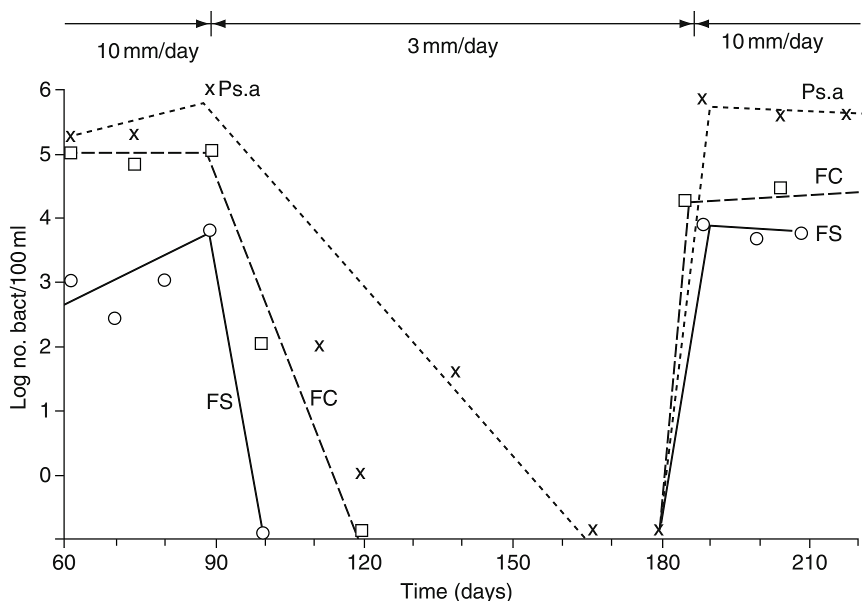


Figure 4 Purification of septic tank effluent spiked with high concentrations of FC (*Fecal Coliform*); FS (*Fecal Streptococcus*) and PSa (*Pseudomas Aeruginosa*) after infiltration into an undisturbed 60 cm long sample of a well structured B horizon in a silt loam soil. Two flow rates produce different purification as the high flux moves around and the low flux moves through the peds (Bouma, 1979).

soil volume directly in touch with the percolating solution was $<0.1\%$ (Fig. 5). Such phenomena are still more evident in unsaturated soils with bypass flow. Field observations of certain chemicals occurring in groundwater are common while models suggest this to be impossible. Soil scientists can make a major contribution to the study of environmental problems by better defining flow patterns in soils and occurrence of hot-spots, ideally combining expertise of pedologists describing soils in the field and physicists knowing how to express such observations in quantitative terms allowing expressions in dynamic simulation models. This is the basis for hydropedology, a new branch of soil science research (e.g., Lin *et al.*, 2006).

4.2.3. Function 3: Providing habitat and gene pool

Habitat describes the living environment for plants and animals, which is strongly determined by climatic factors in terms of temperature, rainfall, and radiation. But also soil moisture contents in space and time and solute composition are important factors and remarks made for functions 1 and 2 apply here as well, offering again an ideal topic for hydropedology. The mentioning of the gene pool in the EU Soil Protection Strategy is interesting as it opens new and as yet hardly explored research opportunities. The DNA of soil organic matter has been subject to evolution for millions of years and experts assume that this can be the source of new medicines in future.

4.2.4. Function 4: Providing the physical and cultural environment for mankind

This function is operational at landscape level and acknowledges the importance of the physical environment for societal development and wellbeing over the centuries. Before the agricultural revolution, introducing chemical

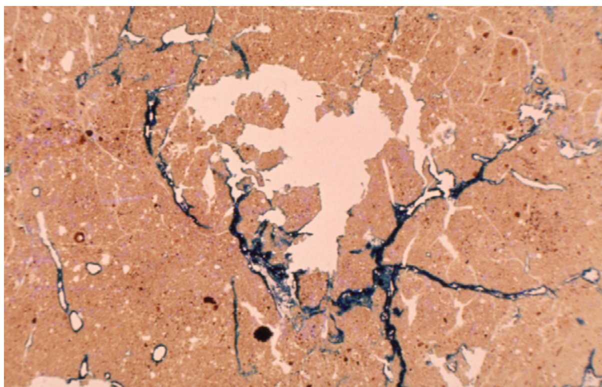


Figure 5 Horizontal section through a structured clay soil after saturated flow. Flow patterns are indicated with methylene blue dye and show that flow only occurs in part of the soil matrix along parts of macropores as a function of macopore-continuity.

fertilizers, drainage, and irrigation, land use was strongly determined by natural, dynamic land and water conditions. In relatively fertile river deltas of the world, settlements, and arable land were established at the higher locations while meadows occupied the lower, wetter areas. Roads and waterways followed natural patterns in the landscape. In upland areas settlements were often associated with sediments or rocks containing relatively high nutrient contents while the availability of water was always a concern and a governing factor for establishing settlements. Modern agriculture has introduced many changes. Farm sizes increased, nutrient management was focused on the needs of agricultural corps or grassland and water management was controlled. This has dramatically changed conditions in many areas. Currently, efforts are being made to preserve some characteristic traditional landscapes as part of the general cultural heritage. These are called: "National Landscapes" in the Netherlands of which 22 have been designated. Farmers receive payments for maintaining characteristic landscape elements, such as hedges. The general policy on National Landscapes does not advocate simple conservation but promotes: "preservation by development." In other words, any possible development should be such that the unique character of the landscape is preserved. This often implies that the water and nutrient regime of an area can only be changed to a limited extent. Different regions have different regimes with corresponding vegetations.

4.2.5. Function 5: Source of raw material

This function is quite divers, varying from excavating subsoil gravel to peat and sod for gardens and greens. Existing detailed soil surveys provide a good guide as to where certain deposits or vegetations occur. Pedologists can delineate certain areas in the signaling phase of the policy cycle, followed in the design phase by formulating alternative future land use scenario's including different quantities and forms of excavation. In former soil surveys, areas were identified as being more or less suitable for excavation, considering technical and availability aspects only. Now excavations have to be viewed in a much broader societal context also considering environmental, ethical, and esthetical aspects. The procedure, deriving land use options, to be illustrated in Figure 6 offers a suitable framework to address this new approach.

4.2.6. Function 6: Acting as a carbon pool

As the climate change discussion continues, there is increasing evidence that soils can be a major sink for Carbon as certain forms of land management can result in a significant increase of the organic matter content. Estimates indicate that carbon storage in soil organic matter could contribute to 1.2 Gt Carbon at world level, which is quite significant (Lal and Follet, 2009; Hillel and Rosenzweig, 2009). Dutch research has shown that organic matter

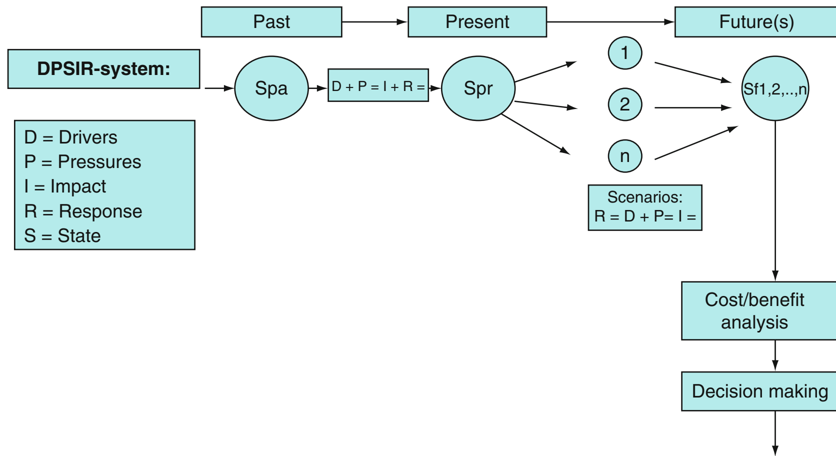


Figure 6 Schematic diagram representing development of future land use scenarios, applying the DPSIR system.

contents in a given soil type can vary significantly as a function of soil management. Droogers and Bouma (1997) coined the term *phenoform* to express management-induced changes in organic matter content of a given soil series, the *genoform*. They showed for a prime agricultural soil in the Netherlands that organic matter contents varied from 5.0 %, 3.3% to 1.7% when using the land for, respectively, permanent grassland, organic farming en high-tech arable. Bulk densities increased in the same sequence from 1.38 kg/m^{-3}, to 1.47 kg/m^{-3} and 1.68 kg/m^{-3}. Soil classifications were identical for the three phenoforms (mixed, mesic Typic Fluvaquent, according to Soil Survey Staff, 1998). A comparable study was made for the most common sandy soil in the Netherlands (a coarse loamy, siliceous, mesic Plagganthreptic Alorthod, according to Soil Survey Staff, 1998, again applying to all phenoforms), by Sonneveld *et al.* (2002). Characteristic land uses were permanent grassland, reseeded grassland and high-tech arable. Organic matter contents were 8.1%, 6.3%, and 4.8% respectively with corresponding bulk densities of 1.30, 1.36, and 1.48 kg/m^{-3}. In both studies a simple regression equation was derived relating the organic matter content of topsoil to past land use with high correlation coefficients of 0.8. This offers the opportunity to go back to mapped areas of a given soil type and sample different fields with known management histories and use equations obtained to predict organic matter contents in future assuming certain types of management. Each soil type is bound to produce different relationships and these could be important for future soil characterization programs.

But the organic matter content is also important to improve the agricultural production potential as a higher organic matter content is associated with a

higher water holding capacity and stronger adsorption of nutrients. Increasing the organic matter content of agricultural soils can therefore also be a contribution towards global food security and soil function 1 (Lal and Follet, 2009).

4.2.7. Function 7: Archive of geological and archeological heritage

This last function has a similar origin as function 4 as the geological and archeological heritage can be seen as part of the physical and cultural environment for humankind. Functions 4 and 7 are different in that function 4 can best be approached by the: ''preservation by development'' concept, while ''preservation'' as such applies to function 7, be it that somehow typical geological and archeological features have to be integrated into the landscape in a satisfactory manner.



5. How to Better Interact with Stakeholders and Policy Makers

5.1. Introduction

Soil science suffers from ''atomization,'' caused by rather separate and isolated activities of different subdisciplines. This inhibits effective interaction with stakeholders and policy makers because only an integrated approach, combining physical, chemical, biological, and space–time techniques, can fully demonstrate soil science´s potential to contribute to solving today's environmental problems as mentioned in Section 3 above. Integration within soil science is therefore a key recommendation of NRC (2009) supported by an additional suggestion to establish Observatories at representative locations where integrated measurements of physical, chemical, and biological soil processes can be made with new techniques. In addition, four aspects will be considered in this chapter that, we believe, have the potential to improve interaction with stakeholders and policy makers: (i) creating awareness for the policy cycle as a basis for research planning; (ii) providing a range of ''options to choose from'' when dealing with certain sustainability problems rather than offering straightforward ''solutions''; (iii) following the ''knowledge chain'' when mobilizing knowledge; and (iv) improving communication.

5.2. The policy cycle

When dealing with the policy arena it is important to be aware of their internal mechanisms. Policy formulation is not a single event but involves a complex trajectory in which the following functions may be distinguished (as cited in Bouma *et al.*, 2007):

1. The *signaling* function, in which problems are identified, preferably based on characterization of actual conditions. Increasingly, signaling involves active engagement of various stakeholders often with strong and deviating opinions.
2. The *design* function in which solutions or alternative options for possible corrective action are defined based on research using existing information or results from new research initiated in the context of the particular problem being studied.
3. The *decision* function in which a selection is made by policymakers or stakeholders of one particular solution presented in phase 2. The decision process often involves intense negotiations during the political decision process.
4. The *implementation* function in which the selected solution to the problem being considered is implemented in practice.
5. The *evaluation* function in which the entire process is evaluated in terms of learning experiences. This function may require observations and monitoring to document results obtained.

To be effective, the role of research in each of these phases should be different and researchers are well advised to be aware of this. *Signaling* can originate from different sources, for example, from various stakeholders, government agencies or action groups. And certainly researchers can play an important role in signaling particular problems, preferably even before they are likely to occur. The interactive character of policy formulation these days requires a broad signaling process to the effect that a wide range of opinions is represented, even ones that may appear rather weird at first sight. Researchers can play an important and active role here in gathering opinions and in providing supporting evidence. *Designing* is the classical role of research and many research projects are confined to this activity. *Deciding* is the prerogative of policymakers or responsible stakeholders. The role of researchers is very limited here even though they sometimes play an important role in feeding the decision process with crucial information at the right time and the right place. *Implementing* is usually not seen as part of the responsibility of researchers. They have performed their research, a decision has been made to implement the results, and it is up to others to follow up. Still, researchers can play an important role in supporting the implementation process by making sure that deviations from the original plans do not occur. The time for new and interesting ideas that are likely to throw implementation off course, is over when implementing. Researchers will have to realize that a measurable success of their work is only reached when at least some of their research results are successfully implemented in the end. If research results too often wind up in a dusty drawer, their reputation is bound to suffer. Being involved in the implementation process expresses therefore well understood self interest. Finally, *evaluating* the entire process

from signaling to implementation is very useful for all participants involved as a joint learning experience.

Most current research projects focus on *design* alone and a plea can be made to broaden the approach to the entire policy cycle, very much including *evaluation*. This will result not only in larger and more costly projects but also in more effective and successful research in the end. Of course, the policy cycle, as represented here, is a strong simplification of reality where different functions sometimes occur simultaneously. Still, distinction of the functions may help to clarify and provide a systematic approach towards defining the role of research in society (see examples in Bouma *et al*., 2007).

5.2.1. Defining options rather than solutions

The scheme of the KNAW (2008), intended to structure soil research transparently and in a form that would effectively feed into the EU sustainability debate, consists of two diagrams, of which only one will be shown here (Fig. 6). The DPSIR scheme is used as in ECSSS (2004) (see also Section 3). Drivers of land use change (D) and the associated Pressures (P) have had an impact in the past, leading to Responses (R), all together resulting in a current State (Spr) of the land, be it a field, farm, region, or country. The State (S) is expressed in terms of the seven soil qualities, discussed above. The present State (Spr) makes a link with the past (Spa), but particularly with possible future States (Sf). For the future, a series of planning options have to be compared, because there is, of course, no single land use option that represents the one and only sustainable land use plan. It is important to represent all ideas when defining options, even the improbable ones to allow a comprehensive analysis in the signaling phase of the policy cycle. Different interests are at stake in each option and they have to be balanced, requiring a negotiating process. Each option is the result of certain tradeoffs between economic, environmental, and social demands. Many of such tradeoffs are "hidden" and researchers have an important task to make them explicit, creating a more transparent process. Production of maps for each option, showing "what" would occur "where" in the area being considered when a given option is followed is a powerful tool in the negotiation process as part of the design phase in the policy cycle. Each option is to be subjected to a cost–benefit analysis. Note that this economic analysis is suggested at the end of the process and not at the beginning, which is common now as economic analyses are often dominant when making land use decisions, ignoring, or de-emphasizing environmental and social aspects. Once decisions have been made, research can still be helpful in the implementation phase of the policy cycle to check whether drivers, pressures, and impacts of a chosen option indeed materialize as foreseen.

A (partial) example of the proposed procedure for an area in the Netherlands was presented by Bouma *et al*. (2008). Issues mentioned in the five

clusters and many of the 110 research priorities of ECSSS (2004), as discussed in Section 3, are also mentioned here when defining the options for a given land area. However, in our approach they are fine-tuned for a given area rather than being generic, while they follow a logical operational sequence rather than an arbitrary listing by threat. Later it will be important to collect and analyze experiences obtained at different locations as part of the evaluation phase of the policy cycle.

Scientists have a special role in the process defined above:

(i) They assemble a range of representative options as articulated by various-stakeholder groups and they add their own. The latter is important because as Henry Ford noticed: "If I would have asked my customers as to what they wanted, I would have built a better horse and buggy." Finding out what stakeholders really want is not a matter of simply asking questions or of sending a questionnaire. The key approach is to find out: "what's in it for me?" and this requires a personal talk, often over a drink. This is a time consuming activity and assistance by communication experts or social scientists is often needed. Some speak of: "knowledge brokers." However, the challenge to obtain a representative range of options for a given problem should not be completely delegated to third parties because meaningful results will only materialize at the end when the various interests are well represented and this requires direct involvement and commitment by the scientists.

(ii) They make estimates by means of computer simulations for each option, if adequate data are available, in terms of its environmental, economic, and social implications and present this to the decision makers, be it, stakeholders or politicians.

So rather than just answering specific and isolated questions by either stakeholders or policy makers, scientists (in this case soil scientists) create a transparent picture of all possible options with internal tradeoffs. This, ideally, will facilitate the negotiation process and avoids scientists being seen as, either spokesmen for certain stakeholders or for policy makers. Thus the important scientific independence is established. This is, of course, a highly idealized picture but it may serve as a general guideline for defining modern relations between soil science and society when the debate focuses on sustainable development, which it often does.

5.3. The knowledge chain

Knowledge is the major tool (or the major weapon) of science. Different forms of knowledge are operational in the policy cycle, discussed above. Current emphasis in research on publishing articles in international, refereed scientific Journals often implies a focus on one particular type of quantitative and mechanistic knowledge. But other types of knowledge are also quite

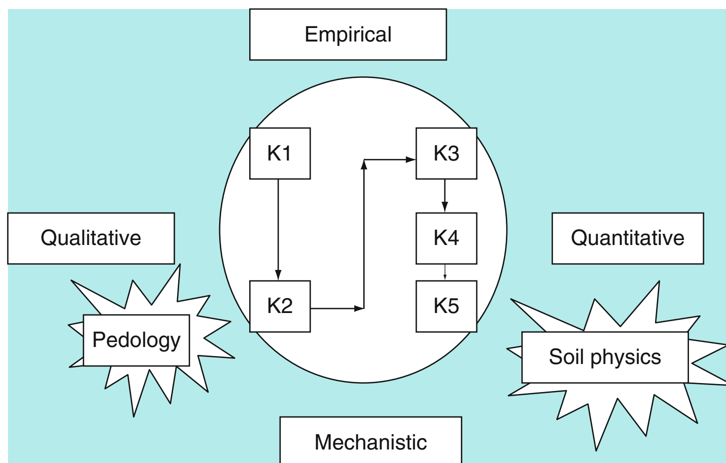


Figure 7 Knowledge diagram, defining five types of knowledge as a function of four criteria, as indicated. Soil survey activities (K2) and soil physical research (K5) combine into hydropedology.

important. The scheme of Fig. 7 (Bouma *et al.*, 2008) defines five types of knowledge (K1–K5) in terms of two ranges of characteristics: from qualitative to quantitative and from empirical to mechanistic. K1 represents user expertise, sometimes referred to as *tacit knowledge;* K2 is expert knowledge from practitioners or applied scientists. The still essentially empirical character of this knowledge is supported by a better understanding of the underlying processes and can be highly effective in practice. K3–K5 represents increasingly specific scientific knowledge. K3 represents, for example, empirical statistical relationships, while the underlying processes are increasingly expressed in quantitative terms in K4 and K5, the latter representing cutting-edge science, a favorite entry point for the scientific journals mentioned above. *Signaling*, as discussed above, requires acknowledgement of K1, K2, and, ever more frequent, K3 knowledge of stakeholders who have access to internet and are increasingly well informed.

At this point in time, K5 knowledge dominates soil science literature and scientific conferences to the extent that its function and place in the sustainability debate and its role in facing the important environmental research issues of today (Section 3) remains unclear. "There are many pieces of the puzzle; how do they fit together and—by the way—is anyone completing the puzzle?" Bouma (2001) made a plea for a sequential approach to research by considering a research chain K1–K5 (Fig. 7). First, find out what the stakeholders know (K1), and then add the experts' opinion (K2). Scientists come in next, explaining what might be gained

by introducing K3 research and, if that does not produce satisfactory results, K4 and, perhaps new K5 research. A cost/benefit analysis is needed to support selection of the proper procedure. One could call this the step-by-step "salesman" principle: rather than suggest a costly K5 research program right away (which is not uncommon these days), the user is introduced stepwise to ever more detailed research. This way, the user is effectively pulled into the research process and becomes part of it. If he is just presented with a (K-5) plan, the reaction is bound to be different. There are, in fact, many projects that can be solved with K3 or K4 expertise. Bouma and Droogers (1999) presented an example of the step-by-step approach by characterizing the soil moisture supply capacity of a particular soil.

Consideration of the knowledge chain is also important for the interaction with policy makers. A question often heard is: "why more research as agricultural research has been in progress for at least a century. What have you guys been doing?" This question is relevant because many research proposals read as if researchers have to start from scratch. It would be wiser to systematically analyze step-by-step what has been done so far and propose new research only after defining what knowledge is missing to reach a given objective.

Traditionally, soil survey (K2) activities were associated with extensive contacts between surveyors and land users. Now that soil mapping is complete, such contacts are gone and, also because of budgetary considerations, less time is available for field work and monitoring and more time is spent behind the computer screen. Hydropedology combines K2 soil survey experience with K5 cutting-edge soil hydrology/soil physics expertise and this is an attractive combination. But systematically following the entire knowledge chain in each subdiscipline is also important as it will cement interaction with stakeholders who will experience being part of a joint learning process rather than being simply a recipient of the scientist's assumed wisdom.

5.4. Improving communication and public relations

5.4.1. Introduction

Efforts are made nowadays to communicate better the importance of soils to funding agencies and the public at large and many of these efforts are successful (e.g., Hartemink and Mc Bratney, 2008). More can be done to improve communication, however, and three possible activities will be discussed here: (i) making the SQ concept operational; (ii) explore the potential of the ecological footprint concept to illustrate the importance of soils for sustainable development, and (iii) write storylines for major soil types, not only covering soil genesis but also their potential in performing soil functions.

5.4.2. Soil quality

Quality measures for water and air are widely accepted and used. Unfortunately, even though many scientific papers have addressed the topic (e.g., Letey *et al*., 2003), there still is not a widely accepted measure for SQ and this presents a clear barrier to effective communication. Bouma (2002) made an attempt to fill the gap and proposed a SQ measure based on soil function 1, as follows:

(Water-limited yield/potential yield) × 100, where calculations are made for a standardized crop. Potential production is determined by climatic data and assumes availability of sufficient water and nutrients and no pests or diseases. The water-limited yield, in contrast, is based on the actual soil water regimes while other assumptions are the same. Also an absolute SQ was defined, based on the maximum potential yield to be achieved anywhere in the world: 41 Mg/ha/year. The SQ concept can also be used to express the effect of different forms of soil degradation, such as erosion, compaction, and sealing by modifying model parameters such as rooting depth, topsoil properties, etc. Three examples are presented in Table 1. Differences in calculated potential production result from different climatic conditions. The SQ (water-limited) of the Zambia soil is relatively low, reflecting a low water supply capacity caused by low rainfall and supply by the soil. SQ (absolute) values are relatively low for all soils as values tend to be higher for moderate climate zones. Effects of erosion expressed in terms of loss of topsoil are relatively strong for the China soil. Thus, comparing different SQ values will allow comparisons among different soils and general estimates of the relative importance of different forms of degradation as well as soil improvements (e.g., increasing the organic matter content or deepening the root-zone).

Table 1 Calculated soil quality (SQ) indicators for three soils as discussed in text (from Bouma, 2002)

	Pot. Prod. (Mg/ha)	SQ (water-lim)	SQ (abs.)	SQ (erosion)
Ferric luvisol (Nigeria)	14	90	32	75
Orthic acrisol (China)	8	90	20	50
Orthic ferralsol (Zambia)	23	50	27	40

5.4.3. The ecological footprint

The ecological footprint is the area of land (in ha's) needed for a certain level of consumption. The footprint can be estimated for a person, a country, a continent or for the world as a whole. The footprint concept is widely publicized on the internet and is a powerful metaphor for the manner in which mankind uses its land and should, therefore, be attractive for soil scientists to illustrate effects of different types of management on different soils. As is, the footprint concept is not used to its full potential. For instance, the available footprint for each world citizen is reported as being: 2.1 ha; for the USA: 9.4 ha; the EU: 4.7 ha and for India: 0.9 ha. For developed countries, more than 50% of the number is, however, determined by energy use that is compensated by planting trees "elsewhere." The picture will change, therefore, as greener solar energy is generated in future. Also, effects of intensifying agriculture or introducing organic farming on a large scale could be demonstrated using the footprint concept, which, as used, defines "land" in a generalized manner. Soil scientists could improve it by using real soil patterns and the productive potentials of soils as a basis for a more realistic analysis that could be quite helpful in demonstrating how precious our soil resource really is.

5.4.4. Prepare storylines for major soil types

Most citizens have a deep affinity with soils. As Hillel (1991) pointed out, most religions describe how God created man from clay or dust. To increase the visibility of soil science various efforts have been made in the past to show the beauty of soils and to tell their story. For example, in the USA, different States of the Union have "State Soils." In Germany, every year a "Soil-of-the-year" is chosen by the German Soil Science Society and this is widely communicated. In the Netherlands, members of the Dutch Soil Science Society elected the "Most characteristic soil of the country." Another idea would be the preparation of well illustrated storylines for major soil types, not only describing their genesis and properties but also their use potentials in terms of the soil functions and their resilience. Every soil has a characteristic story to tell! This story was of course, told in traditional soil survey reports but individual soils were hidden in a broader context and interpretations were descriptive and normative, rather than quantitative and optional. Two partial Dutch examples of storylines were mentioned above in Section 4.2.6 (Droogers and Bouma, 1997; Sonneveld *et al.*, 2002). Highlighting particular soil types is important as many policy reports refer to soils in a generic manner only, while soil diversity is one of the key assets of the soil resource (see also: Hillel and Rosenzweig, 2009).

6. The Internal Issue: How to Become an Effective Community of Scientific Practice?

6.1. Introduction

The need for scientists to be more in touch with stakeholders was articulated effectively by Wenger *et al.* (2002) when they discussed their concept of ''Communities of Practice'' (CoPs). Scientists work together with various stakeholders and policy makers in CoPs but the way this can best be done has not as yet been well described. There are too many examples of well intentioned scientists getting lost in broad, wide-ranging debates about various aspects of sustainable development. This certainly happened to soil scientists and was a major reason for a plea earlier in this chapter to focus research on the soil functions to be better prepared in terms of tools to be used. However, this first requires soil scientists to get their own act together before they branch out to meet others, be it colleague scientists or stakeholders. The need to avoid further atomization of the soil science profession was articulated by Bouma (2005) who proposed a ''Community of Scientific Practice'' (CSP) as a: ''close-knit interactive team in which scientists do fundamental, basic, strategic or applied research in a chain-mode as needed while others interact within Communities of Practice (CoP's) with stakeholders and policy makers or work on communication or education.''

A soil-CSP could first integrate soil studies performed by various subdisciplines before branching out to other scientists in an interdisciplinary mode and, after that, to CoPs where transdisciplinarity would be required. Within a CSP there are different tasks that are to be fine-tuned to the particular qualities of participating scientists. For example: A good basic scientist should not be forced to work with stakeholders, nor should a more applied scientist be forced to produce a large number of scientific papers. Diverse criteria to judge research are needed for a CSP. The traditional mode-1 criteria still apply to basic scientists but the more applied mode-2 work needs new criteria that have been defined and are available (KNAW, 2005). In addition, three aspects need special attention when discussing soil-CSPs: (i) guarding the scientific quality and integrity of its members; (ii) continuing basic research to make sure that the scientific vitality of the CSP is maintained and (iii) balancing soil education programs to represent the entire knowledge chain by promoting: ''T-shape skills.''

6.2. Scientific quality and integrity

Using accessible databases, pedotransferfunctions, user-friendly simulation models, remote sensing and flashy 3D/GIS systems, anyone who can handle a computer can produce good looking but not necessarily good ''soil

reports." An illustration of this phenomenon was presented in Section 4.2 of this chapter when describing soil functions 1 and 2 where simple representations are being used by non soil scientists of moisture supply and transformation of compounds. Soil scientists have to show that such simplified procedures produce poor and misleading results and they should indicate how to do a better job which may imply the need for new research. In fact, many young graduates start their own consulting firms, without having access to research and when this implies data mining without replenishment we are entering a dead-end street for the profession. Quality control is essential and broad experience obtained in the USA with certification of professionals could be valuable for other countries as well.

6.3. The need for basic research

Most important of all: we still know relatively little about the most complex ecosystem of the world: the soil. Continued basic soil research *that fits in the knowledge chain K1–K5*, is essential to remain a viable and thriving scientific discipline that is recognized by its peers. As stated above, just mining existing data will lead to death of the profession. Working within the knowledge chain can help to focus the research effort without restricting scientific independence. This is needed because soil science meetings and conferences still present a very wide array of isolated, unrelated microtopics and it is hard to see how this could somehow result in meaningful input for the study of the major issues of today, as outlined in Section 3 of this chapter. As we are a relatively small group of scientists, disciplinary scattering of research activities (the: "atomization" of soil science) is a real danger. That is why the suggestion was made earlier in this chapter to provide a major thrust in our research on the soil functions, working in the context of first CSPs and then CoPs.

6.4. Soil education

Educational programs for soil science are being revised everywhere at Universities and Technical Schools reflecting results of new research and developments in information and communication technology. In this context, distance learning becomes increasingly important. Serious attempts are being made to reflect mode-2 thinking but this is difficult because the research field is still struggling with the concept as was illustrated in the previous sections of this chapter. There is a tendency to include social and communication science into soil science curricula which is, as such, a good idea but when this is presented in only a theoretical context it may contribute little to developing an operational mode-2 approach for soil science. Moreover, there is a real risk that not enough time remains for basic courses in soil science itself at K5 level and when students do not learn the basic

sciences in school, they never will and when basic sciences will not feed the profession in future, it will die. There is therefore a need to develop "storylines" for particular problems (see Section 5.4.4 where the storyline was restricted to a given soil type) including case studies built around the central question as to how various types of knowledge can facilitate interaction and communication to the effect that problems are solved in the end. Ideally, students should learn: "T-shaped skills," the vertical line representing disciplinary skills and the horizontal line the ability to interact and communicate effectively. Within a CSP the vertical and horizontal axes may vary in length, depending on the particular abilities of participating scientists, but they all should still form a T ! And, finally, a major challenge for all education efforts is to make students aware of the basic differences between data, information (data in context), knowledge (internalized information) and undefinable wisdom which is the ability to do the right things at the right time or, quite often, to not do anything at all for the time being.

7. Conclusions

- The Knowledge Paradox—research results do not sufficiently lead to application and innovation in society—certainly applies to soil science. More interaction with stakeholders and policy makers is widely acknowledged to be necessary, certainly by funding agencies. The soil science community is, however, still rather confused as to how to proceed because broadening its range of activities could be harmful to the scientific integrity of the profession. But action is needed as science, and certainly soil science, has to recover its "societal license to research."
- Realize that you can only be a serious partner in a transdisciplinary team (where scientists from different disciplines interact with stakeholders and policy makers in Communities of Practice, CoPs) when you have unique contributions to make studying sustainable development. Define your soil-science: "niche."
- Do not directly focus on soil threats, which are very broad including socio-economics, but study first the soil functions (our "tools," filling our: "niche") taking results later to the larger transdisciplinary groups studying current and future threats.
- Try to get the internal soil-science "act" together: current atomization of subdisciplines and the separation of field, lab, and computer is detrimental for the profession. Suggestions are to: (i) reestablish the knowledge chain (K1–K5), linking tacit knowledge all the way to cutting-edge expertise and work jointly on the functions and their application, and (ii) be aware of the policy cycle when planning and executing research.

- Consider in the above context the concept of a CSP for soil science where the complete knowledge chain (K1–K5) is represented as well as specialists in communication and interaction. Feed experiences obtained in CSPs into CoPs and educational programs.
- Pursue basic research that is connected to the knowledge chain by following a step-by-step approach to research planning. Without basic research, mining only of existing data will mean the untimely death of a scientifically vital soil science profession. An array of new observation, monitoring, and measuring techniques is ready to be applied and is needed to open new frontiers.
- Vigorously protect the independence of researchers by not being ''his-master's-voice'' of financers, stakeholder groups, NGOs, or policy makers. Go, if possible, for defining ''options'' to choose from rather than single ''solutions'' to problems and, above all, create science-based transparency for decision making!
- Build on the fascination with soils of the public at large by improving communication about soils by developing, for example, operational SQ and footprint concepts as well as exciting storylines for major soil types.

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CHAPTER FIVE

Grassland Fire Management in Future Climate

A. L. Sullivan

Contents

Abstract

A thorough understanding of the behavior of fire in grasslands is critical to the minimization of the impact of fires on agricultural and pastoral land as well as the successful management of the health, robustness, and species diversity of native grasslands. This is also necessary to understand the impact that a changing climate will have on these fires and the subsequent impacts and adaptation steps needed to protect valuable farmland and grassland

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ecosystems in the future, a challenge that will soon be facing all land managers. While a number of studies have investigated the impact of climate change on fire danger indices, the fire danger systems used in Australia are actually fire weather indices that provide no information about the likely impact of climate change on fire behavior.

This chapter summarizes the state of the knowledge of fire behavior in grass fuels and discusses in detail the factors that influence the behavior of grassfires. The CSIRO Grassland Fire Spread Meter is the recommended operational system for the prediction of grassfire behavior in all Australian grass types. The system is used to assess the impact of a high-emission climate change scenario upon the likely behavior of grassfires throughout the fire season for three major pastoral and agricultural regions of eastern Australia in 2020 and 2050. It was found that mean fire rate of forward spread in ungrazed/natural pastures will increase by a maximum of 10% by 2020 and by 32% by 2050 in southeastern Australia. The implications for grassland management strategies and possible climate adaptation pathways are explored.

1. Introduction

In Australia, a continent in excess of 7.6 million km^2 in area of which 61% is considered temperate and 39% tropical, over three-quarters of the land area is grassland of one type or the other (Moore, 1993). While much of the country is arid or semiarid, over half of the total land area (58% or 442.8 Mha in 2006) has been established for agricultural purposes (ABS, 2008). The majority of farmland is used for grazing sheep (for meat or wool) and cattle (for meat or milk). In the tropical and subtropical north much of the cattle grazing is carried out on natural[1] open grasslands and woodlands or open forests with a grassy understorey. In more temperate zones, grazing of sheep and cattle is carried out in improved pastures, often cleared of trees, consisting primarily of introduced annual and perennial species for forage crops, including *Phalaris* spp. and subterranean clover (*Trifolium subterraneum*).

In 2007/08, 24.1 Mha of the farmland area was cropped with a variety of cereal species; more than half of this area was planted with wheat (Crooks *et al.*, 2009), predominantly in the temperate regions of southeastern Queensland, central New South Wales and Victoria, southeastern South Australia and southern Western Australia. Less than 1% of the farmland area is

[1] Moore (1970) defined three categories of grasslands in Australia: native (communities comprised solely of native species); natural (comprising mostly native species but with a high proportion of volunteer introduced species; and improved pastures (communities comprised entirely or principally of sown species to which fertilizer is often applied).

irrigated, primarily for grazing (ABS, 2008). Over 90% of the cereals cropped in Australia are grown during the Winter (Autumn–Winter–Spring) season. These include wheat, oats, barley, and rye, which are often grown in rotation with pastures such as subterranean clover, medics, or lucerne and with other winter crops such as canola, field peas and lupins. Summer (Spring–Summer–Autumn) cropping generally consists of sorghum grain, rice, and maize, which may be grown in rotation with winter cereals in some areas.

In 2007/08, the rural farming sector had a gross value of production of $41.2B (~2.3% gross domestic product) with $27B in export value (Crooks *et al.*, 2009). Protecting the food and livestock produced and the infrastructure necessary to produce it in the rural sector is a primary requirement to ensure food security in the future. While climate change will pose many challenges in ensuring security of Australia's food future, from pests and diseases to changed productivity, the risk posed by bushfire is one that is endemic in the Australian landscape and has the potential to affect far more than just this year's food production and exports. This may have increased impact in the future as attention turns to growing feedstock for biofuels for renewable energy. On average, annually, bushfires cost Australia in the order of $77M (BTE, 2001); however, actual annual costs depend on the severity of the fire season and recent years have seen much more significant costs related to high-impact bushfire events (e.g., Ash Wednesday, February 16, 1983 ($324M) (Blong, 2005); January 18, 2003 ($350M) (Blong, 2005), February 7, 2009 ($1.02B)[2]). Primary rural losses generally occur in terms of livestock, fencing and feedstock with sometimes significant collateral damage in terms of people's lives, livelihoods, and infrastructure such as buildings and powerlines as well as reduced ecosystem services such as water quality. Understanding the threat posed by bushfires in the rural sector is an important step in reducing the risk to rural landholdings, livestock, infrastructure, and lives.

1.1. Fire in Australia

Fire has long been a defining factor in Australia's history. Many of the native species of flora in Australia have evolved with recurrence of fire in the landscape (Kemp, 1981; Smith, 1979; Springett, 1979) and, indeed, in many cases require fire to propagate (Bradstock *et al.*, 2002; Gill, 1981; Kemp, 1981; Pyne, 1991). Australia's weather, climate, and topography and native vegetation are conducive to the occurrence of widespread fires. The fire season in Australia, that period when bushfires are likely to occur, shifts from mid-Winter to mid-Spring in the north to late Summer to late Autumn in the south with occasional serious fires occurring to either side

[2] Insurance Council of Australia estimate, published on Emergency Management Australia website (http://www.ema.gov.au), accessed 5/8/2009.

of these periods depending on antecedent conditions (Luke and McArthur, 1978). Each year, an average of 53 Mha is burnt by fire (Russell-Smith *et al.*, 2007), of which it is estimated that about 2.5 Mha is burnt through unplanned occurrence of fire (i.e., wildfires) (Cheney and Sullivan, 2008). In some years, particularly following good growth, grasses can form a continuous sward through the arid and semiarid regions of central Australia and may lead to the burning of exceptionally large areas (Luke and McArthur, 1978). In the north, where agricultural and cultural burning is common, very large areas may be burnt as these fires are often allowed to spread unchecked for extended periods (Russell-Smith *et al.*, 2007).

Fire is often used in management of forested lands for species regeneration purposes and removal of postlogging residue (Chandler *et al.*, 1983) as well as for reducing the hazard posed by accumulating understorey fuel (McArthur, 1962, 1965). In the tropics, subtropics, arid, semiarid, and temperate regions, fire is also used for native grassland management across sectors, for example in pastoral, mining, conservation, Indigenous lands, military, etc. (e.g., Andersen *et al.*, 2005; Burrows *et al.*, 1991; Griffin *et al.*, 1983; Marsden-Smedley and Catchpole, 1995; Noble and Vines, 1993; Williams *et al.*, 1998, 2003). In agricultural lands, fire has traditionally been used in the removal of postharvest residue and invasive weed management. However, conservation farming practices such as direct drill sowing, minimum tillage and postharvest stubble retention are increasingly being employed to reduce effects of nutrient and moisture loss, erosion, etc. (Adcock, 2005) and to allow for cropping in continuous cycles with minimum degradation of soil properties (Chan and Heenan, 1996). This has the effect of reducing the extent to which fire is used as a tool as well as the reduction in the extent of grazing outside of the winter growing season. It also has the effect of changing the structure, amount, and continuity of residual fuel present in paddocks after harvest and thus increasing the potential for more extensive wildfires than under previous management regimes (Tolhurst *et al.*, 2008a).

While much of the perceived impact of bushfires occurs from fires burning in forested areas, fires burning in grassy fuels can spread rapidly under conducive conditions, burnout considerable areas in a short amount of time, and pose a significant threat some distance from the point of origin. In Australia, fires are often a single day event, especially in grassy fuels, and suppression efforts often cannot be organized quickly enough to have an impact on the spread and behavior before serious damage occurs.

1.2. Fire management systems

To manage fire in the landscape one must have a comprehensive understanding of the behavior of fires and the factors that influence that behavior. Fire behavior is a term to describe the general characteristics of a fire such as

its rate of forward spread in the direction of the wind, its fire-line intensity, the flame geometry (Gill and Knight, 1991), rate of perimeter growth, etc. The study of fire behavior has been underway in a variety of fuels around the world since the early twentieth century. The study of fire behavior in the grassland did not initially attract the same attention as the study of large forest fires (occurrences of which often made international headlines) but the economic importance of the impact of large widespread and fast-moving fires in agricultural zones demanded attention.

Numerous systems have been developed around the world to predict the behavior of fires in grassy fuels. These include BEHAVE in the USA (Andrews, 1986; Burgan and Rothermel, 1984; Rothermel, 1972), the Canadian Fire Behaviour Prediction (CFBP) system in Canada, and the Commonwealth Scientific and Industrial Research Organisation (CSIRO) Grassland Fire Spread Meter (GFSM) (Cheney *et al.*, 1998; CSIRO, 1997b) in Australia. However, the BEHAVE and CFBP systems were designed to be generally applicable to a range of fuel types utilizing standardized fuel parameters to specify a particular fuel's attributes. The CSIRO GFSM, on the other hand, was designed specifically for Australian grass fuels and is applicable to both native and exotic grasslands. It replaced the spread prediction system developed by McArthur (1966), although McArthur's method of calculating fire danger (a measure of the potential for fires to start and then cause damage; Chandler *et al.*, 1983) was retained (CSIRO, 1997c). The CSIRO GFSM has been in use by all Australian rural fire authorities for all types of grass fuels since its introduction in 1997. This system represents the state of the art of fire spread prediction in grassy fuels and allows not only the prediction of the movement of the fastest part of the fire perimeter but, in conjunction with a fire spread simulation component, can be used to simulate the spread of the entire perimeter across the landscape (Coleman and Sullivan, 1996; Tolhurst *et al.*, 2008b).

1.3. Fire and climate change

Recent recognition of the potential impact that changed climate due to global warming will have on ecosystem management has raised concerns about natural resource management around the world. Lucas *et al.* (2007) found that in the grasslands of southeastern Australia, a region projected to become hotter and drier under forecast climate change, the number of days of extreme fire danger will increase by around 5–30% by 2020. Similarly, Pitman *et al.* (2007) found that the probability of extreme fire danger in eastern Australia will increase by 25% by 2050, regardless of the greenhouse gas emission scenario, and by an additional 20% by 2100 under a low emission scenario and by 50–100% under a high emission scenario.

While grassland fire danger is important for determining the likelihood of ignition, preparation of public warnings, suppression readiness of

firefighters and the likely difficulty of suppression of a fire burning under those conditions, it does not necessarily reveal information about the likely behavior of a fire burning under those conditions. Cheney and Gould (1995b) found that the factors used to determine fire danger interacted in slightly different ways than those used to determine fire behavior.

To understand the likely behavior of a grassfire under changed climate requires an understanding of fire behavior in general. The purpose of this chapter is to aid land managers charged with management of both natural and nonnatural grasslands in Australia and elsewhere to gain a better understanding of grassland fire behavior in the landscape and the tools used to predict that behavior now and under possible future climate conditions. The constructions of the CSIRO GFSM and Fire Spread Meter for Northern Australia (FSMNA) are presented in detail and comparisons between the prediction of fire spread and fire danger made. The GFSM is then used to assess the likely changes in rate of spread under the same climate scenarios assessed by Lucas *et al.* (2007) for fire danger. Implications for native grassland fire management strategies under changed climate are then discussed and recommendations for adaptation pathways given.

2. Fire Behavior in Grasslands

This section and the one following provide an overview of the behavior and prediction of fire in grasslands. The interested reader is directed to Cheney and Sullivan (2008) for a more comprehensive discussion of these matters.

The type of vegetation in which a fire predominantly burns determines the type of fire. Thus a fire in grassy fuel is called a grassfire, a fire in forest fuel is called a forest fire and a fire in heath fuel is called a heath fire. However, generically a fire in any vegetation type is called a wildland fire in North America or a bushfire in Australia, South Africa and other parts of the world. As such, all bushfires have a number of common traits, regardless of the type of fuel in which they are burning.

All bushfires burn in the direction of the prevailing wind; this direction may be modified by the effects of the local topography. The most active downwind section of the fire perimeter is called the headfire and is characterized by high flames, intense behavior, incomplete combustion, dark smoke, and continuous spread with the wind (called heading spread). The section of the fire burning into the wind is called the back and is characterized by lower flames, less intense behavior, more complete combustion, lighter colored smoke and slower (backing) spread. The parts of the fire perpendicular to the wind are called the flanks and are characterized by

behavior alternating between heading and backing spread as the wind direction shifts due to turbulence in the air flow.

Fire in grass fuel is characterized by its rapid combustion and high rate of spread. The fine, high surface area-to-volume ratio of a grass leaf, its high thermal conductivity, low density, and vertical orientation, mean that a blade of grass will rapidly ignite and just as rapidly burnout. In contrast with forest litter fuels, which are coarse, thermally thick, of higher density, more horizontally oriented and have burnout times in the order of minutes (Burrows, 2001), grass fuels have burnout times in the order of 5–15 s. Also grassfires are generally more open to the wind than forest fuels which are sheltered by the presence of the trees.

As a result of the nature of the fuel and the exposure to the full force of the wind in the open, grassfires are highly responsive to changes in the direction and speed of the wind. Rates of spread of large grassfires burning for extended periods in the open under relatively extreme fire weather conditions can be in the order of 15–20 km h^{-1}. The fastest reliably recorded sustained speed for a grassfire is 27.3 km h^{-1} (Noble, 1991), although speeds in excess of 30 km h^{-1} over shorter intervals have been reported (Gould, 2006). Large forest fires burning for extended periods under the same sort of weather conditions will move at a much more pedestrian 5–10 km h^{-1}, although the effects of spotting (the ignition of fires ahead of the main fire by lofted firebrands) complicates the estimation of the speed of these fires and may allow fire to spread over large distances quite rapidly. Due to its potentially high rate of spread, a grassfire can travel a great distance in a short length of time and cause significant damage a long way from its origin.

The flames of a bushfire are of the turbulent diffusion type, the result of the turbulent mixing of burning volatile gas released from the thermal degradation of the solid fuel and oxygen in the air. The maximum height of the flames is dependent upon the rate of combustion, the height of the fuel and the speed of the wind but in a grassfire is in the order of 3–6 m. Forest fire flames burning through surface fuels are generally 2–10 m but when the fire escalates under extreme conditions into other fuel strata can be 2–3 times the height of the forest canopy.

The height of the flames of a fire may be used to estimate the fireline intensity of the fire. Fireline intensity, I (kW m^{-1}), is a quantification of the rate of energy release per lineal meter of the head of the fire calculated by the product of the heat yield of the fuel per unit mass, H (J kg^{-1}), the mass of fuel consumed per unit area, w (kg m^{-2}), and the rate of forward spread of the fire, R (m s^{-1}) (Byram, 1959). While not useful for comparing fires in different fuels, fireline intensity does provides a useful metric for comparing fires burning under different conditions in the same type of fuel. Grassfires may reach a fireline intensity of 60,000 kW m^{-1} at the head (Cheney and Sullivan, 2008).

Control of a fast-moving grassfire, particularly direct suppression of the head, is extremely difficult. To limit the area burnt, direct suppression of the rear and flanks is essential to limit the lateral spread of the fire and thus the width of the fire. This in turn restricts the rate of forward spread (Cheney and Gould, 1997). Indirect suppression methods such as those used in fighting forest fires (e.g., backburning and bare-earth fireline construction) are generally of less use due to the time required to initiate and execute such actions and the distance a grassfire will travel in that time.

Prediction of the spread of the fire, primarily the rate of spread of the headfire, is a valuable tool in the determination of the likely behavior of an intentionally lit fire for land management purposes as well as the behavior, speed and area likely to be burnt (including possible assets at risk) of wildfires. This information can be used in the planning of control and suppression action and the determination of necessary suppression resources. A number of tools have been developed to carry out the prediction of grassfire behavior around the world. In Australia, the CSIRO grassland fire spread prediction system, encompassed in the GFSM (CSIRO, 1997b) and FSMNA (CSIRO, 1997a) meters are the tools currently recommended for the prediction of fire spread in grass fuels.

3. Fire Spread Prediction in Australian Grasslands

The CSIRO grassland fire spread prediction system was developed from 121 large field-scale experimental fires conducted in the grass plains of the Northern Territory, Australia (Cheney *et al.*, 1993). These experiments were conducted in two grass species representative of improved pasture species in southern Australia in which grass height and load were varied through harvesting treatments. The fires were ignited primarily from lines 30–175 m long and allowed to burn freely over areas ranging in size from 1 to 6 ha. During each experiment behavioral characteristics such as rate of forward spread, flame depth, and head fire width were observed directly or measured from oblique aerial photographs taken at irregular intervals. Fuel characteristics (height, load, curing, moisture content, etc.) were measured on four transects through each plot approximately every 25 m. The wind speed at a height of 2 m measured at the corner of each plot was averaged for each spread observation (i.e., photograph) interval.

An empirical model of rate of fire spread in grass fuel was constructed using statistical regression tempered by physical understanding of threshold behaviors at low wind speeds (Cheney *et al.*, 1998). The most important factors found to influence fire spread in continuous grassy fuels were:

- head fire width,
- fuel condition,
- grass curing state,
- dead fuel moisture content, and
- wind speed.

3.1. Head fire width

The operational system was designed to predict the mean potential rate of forward spread of the head of a fire burning in a uniform and continuous grassy fuel. A fire reaches its potential rate of spread after it has completed its build-up or development phase during which its width, normal to the direction of the prevailing wind has not been constrained. The width of head fire required for a fire to attain its potential rate of spread increases with increasing wind speed (Cheney and Gould, 1995a, 1997). Figure 1 shows

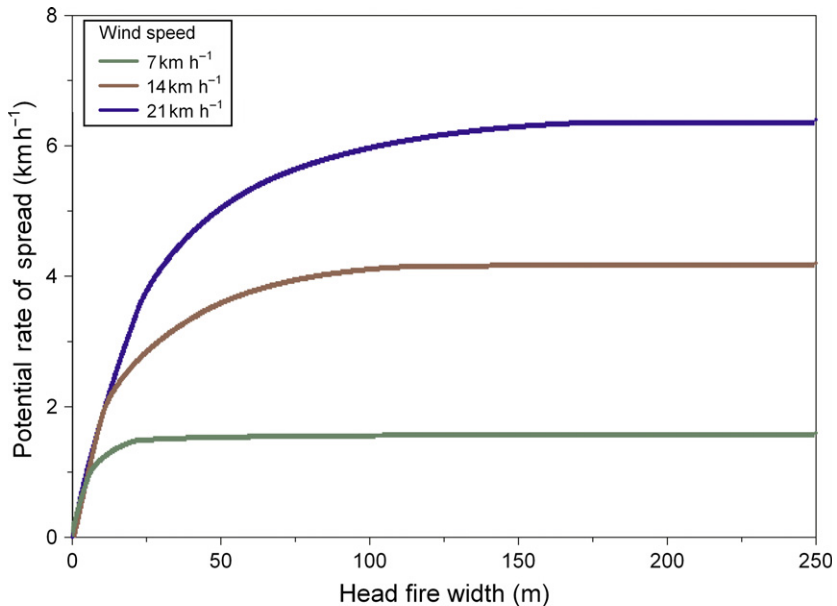


Figure 1 The quasi-steady rate of spread that will be reached for a given wind speed depends on the width of the fire. If the width is constricted, the potential rate of spread will be less than if the width is not constricted. At a low wind speed of 7 km h^{-1}, a fire reaches its potential rate of spread (1.7 km h^{-1}) when the head fire width is around 30 m. At a wind speed of 14 km h^{-1}, the head fire width must exceed 100 m before the fire reaches its potential rate of spread.

the quasi-steady rate of spread that will be reached for a particular width of head fire at wind speeds of 7, 14, and 21 km h^{-1} (Cheney and Sullivan, 2008). At a low wind speed of 7 km h^{-1}, a fire reaches its potential rate of spread (1.7 km h^{-1}) when the head fire width is around 30 m. At a wind speed of 14 km h^{-1}, the head fire width must exceed 100 m before the fire reaches its potential rate of spread, while at a high wind speed of 21 km h^{-1} the head fire width must be greater than 150 m. Under strong winds (i.e., $>$20 km h^{-1}) fires will continue to increase their rate of spread until the head fire reaches a width of around 200 m.

Generally, most fires will reach the potential rate of spread in 30 min, but because the time taken depends on fluctuations in wind direction there are no accurate relationships that can be used to describe the growth of a with time. In the development of the CSIRO grassland fire spread prediction system, Cheney *et al.* (1998) assumed that a fire would not be constrained in its growth and would reach its potential rate of spread. As a result, the system will overpredict the rate of forward spread if the fire has not reached its minimum width for the prevailing conditions; that is it is still in its development phase or its width is constrained in some way.

3.2. Fuel condition

Three distinct fuel conditions in open grasslands have been defined to describe the main types encountered in pastoral, agricultural, open country, woodland, or open forest where the fuel bed is comprised of continuous annual or perennial grasses:

- undisturbed and/or very lightly grazed natural grassland or improved pasture or unharvested crops, generally more than 50 cm tall;
- grazed or mown pasture, generally less than 10 cm tall. This is the most common condition through the agricultural and pastoral zones of southern Australia during summer, and is the pasture condition recommended for predicting the general spread of fire across the landscape;
- very heavily grazed and eaten-out pasture, generally less than 3 cm tall, with scattered patches of bare ground. This condition may be common in southern Australia during severe drought when feedstocks are low.

Fires in closely grazed or mown pastures will spread about 20% slower than those in natural/ungrazed grasses, regardless of the species, fineness, or amount of fuel present, provided the pasture is continuous (Cheney *et al.*, 1993, 1998). The amount of vegetation (i.e., fuel load) by itself was not found to influence rate of spread of grassfires, unlike other models (e.g., Rothermel, 1972). In cases where fuel load directly influenced a fire's rate of spread, Cheney *et al.* (1993) found that this occurred in grasslands only if changes in fuel load also reflected changes in fuel condition, in particular changes in fuel continuity. Heavy fuel loads generally imply swards of tall

standing grass while light fuel loads are normally associated with heavily grazed or even eaten-out pastures in which fuel continuity becomes critical. In these situations it is the condition of the pasture—that is, the degree of grazing—that determines the rate of spread of a fire. Fuel load by itself will influence flame height, flame depth and fire intensity, and thereby the difficulty of suppression.

3.3. Degree of curing

Grass curing is the term used to describe the process of dying and drying out that an annual grass plant undergoes during senescence following flowering and setting of seed. The curing state of a grassland, expressed as the fraction of dead material in the sward, indicates the percentage of the grassland in which the moisture content is dictated by the atmospheric conditions and not moisture of the live cells. This percentage has an important influence on both the ability of a fire to spread across the landscape and its rate of spread (Fig. 2). This function is described by (Cheney *et al.*, 1998):

$$\phi C = 1.12/(1 + 59.2\ \exp[-0.124(C - 50)]) \tag{1}$$

where C is the degree of curing (%) and ϕC is the curing function coefficient.

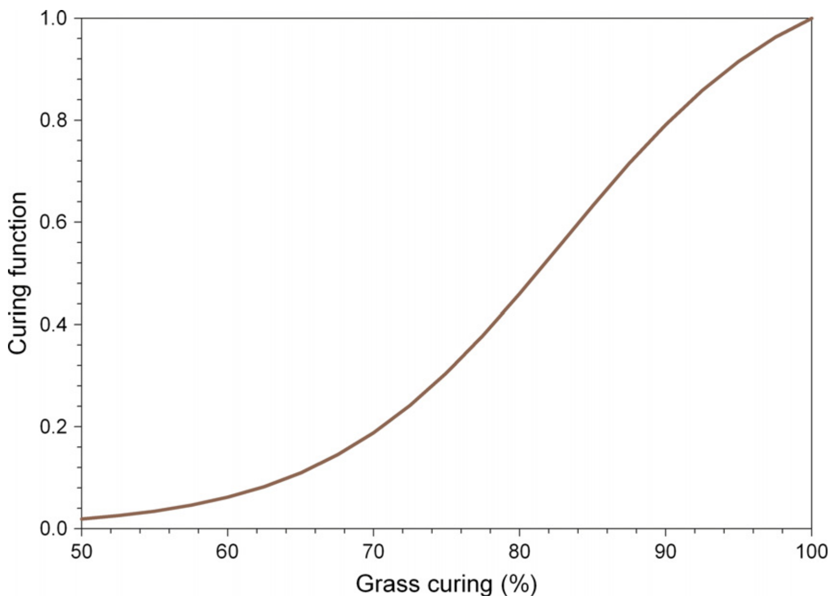


Figure 2 The degree of grass curing has its greatest effect between 70% and 80%. Below 50% fires will not burn. Above 90% fires will spread at near their potential rate of spread.

Grass curing state changes only relatively slowly with time; the plants losing moisture over a period of 6–10 weeks. Once started, the curing process in annual pastures is not affected to any great extent by subsequent rainfall, although if the rainfall is sufficient to germinate seed green shoots may appear beneath the old sward. The process can be rapidly accelerated, perhaps by as much as a week, by a single day of strong, hot, dry winds (McArthur, 1966).

Perennial pastures cure more slowly than annual grasses, and curing is further delayed by rains early in the dry season. As perennials do not need to produce seed to continue their life cycle, rainfall, after curing has started, will delay the curing process in older leaves and produce new green shoots from the base of the clump that will continue to grow.

With practice, it is possible to make a reasonable estimate of curing state by eye and handbooks have been developed to aid this (CFA, 1999), although methods have also been developed to estimate the degree of grass curing using remote sensing techniques (Dilley *et al.*, 2004; Paltridge and Barber, 1985). However, because of spatial differences in curing rates, one should not expect an accurate prediction of fire spread across a particular landscape until all the grass is fully cured. Grass on dry ridges cures more rapidly than that in moist low-lying areas or creek lines. Therefore, early in the dry season fires that spread rapidly through fully cured pastures on ridges will be slowed by green grasses in depressions and stopped where pastures are less than 50% cured.

3.4. Dead fuel moisture content

Dead fuel moisture content expresses the amount of moisture in dead fuel as the percentage of its oven-dry weight. Dead fuel moisture content is not easy to measure directly or to estimate in the field. It is measured experimentally by oven-drying samples of standing grass for 24 h at 103 °C (Cheney *et al.*, 1993; Matthews, 2010). Devices based on changes in chemical or electrical characteristics have been developed for field assessment of fuel moisture content but these often have to be calibrated for particular fuel types (Chatto and Tohurst, 1997). Visual signs provide some guide to fuel moisture content; for example, grasses are limp when they are moist, and leaves become brittle and curl up when they are very dry. However, there is no reliable way of visually estimating moisture content in the relatively narrow range of 2–10% oven-dry weight where a change of 2% or 3% can substantially alter a fire's rate of spread. Models are often used to predict the moisture content of fuels and range from relatively simple correlations of atmospheric variables (McArthur, 1966) to fully physical process-based models (Matthews, 2006).

While the GFSM and FSMNA includes dead fuel moisture content as an independent variable and thus can be measured directly by any means, it also includes the fuel moisture model of McArthur (1966) which correlates air temperature with fuel moisture content through the relative humidity of the air (Fig. 3). This model is described by:

$$M_f = 9.58 - 0.205T + 0.138H \tag{2}$$

where M_f is the dead fuel moisture content (%), T is the air temperature (°C), and H is relative humidity (%) and is shown in Fig. 3.

The temperature and relative humidity of the ambient air should be measured at a standard height of 1.2 m above the ground with wet and dry bulb thermometers in a Stevenson screen, or by using an aspirated psychrometer.

The effect of dead fuel moisture content on rate of forward spread is given in Fig. 4 (Cheney *et al.*, 1998) and is described by:

$$\phi M = \begin{cases} \exp(-0.108M_f), & M_f < 12\% \\ 0.684 - 0.0342M_f, & M_f > 12\%, \quad U_{10} < 10\,\mathrm{km\,h^{-1}} \\ 0.547 - 0.0228M_f, & M_f > 12\%, \quad U_{10} > 10\,\mathrm{km\,h^{-1}} \end{cases} \tag{3}$$

where U_{10} is the wind speed at 10 m in the open (km h^{-1}).

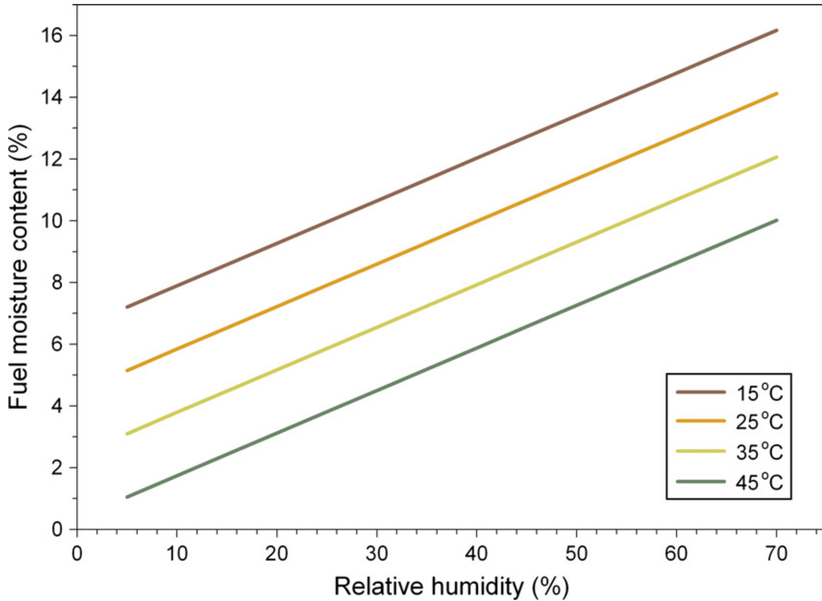


Figure 3 The model of fuel moisture content of McArthur (1966) used in the CSIRO grassland fire spread prediction system.

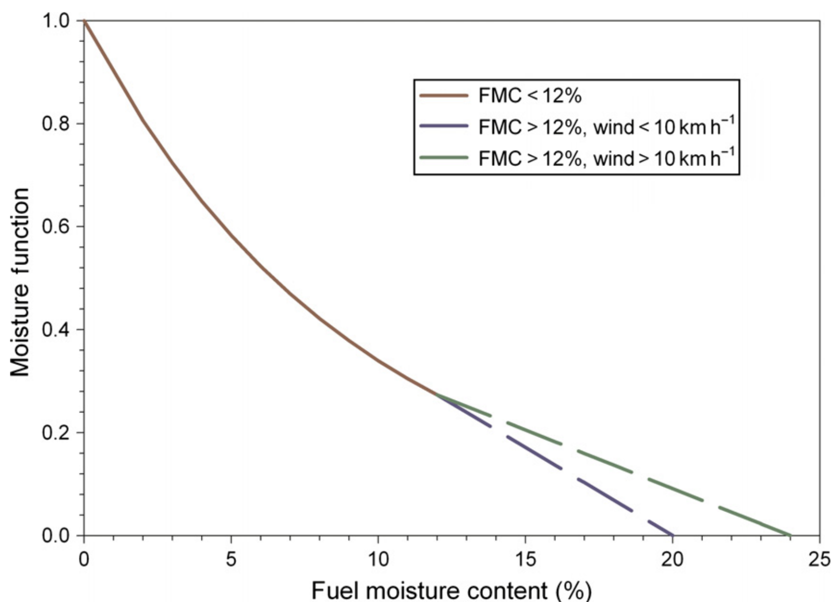


Figure 4 Wind speed in excess of 10 km h^{-1} in the open increases the dead fuel moisture content at which fire will still burn.

Wind speed in excess of 10 km h^{-1} in the open increases the dead fuel moisture content at which fire will still burn. Without wind, the moisture content of extinction (the value at which fire fails to sustain and will cease to spread) is 20%. With wind it is 24%. Table 1 gives descriptions of likely fire behavior at a range of fuel moisture contents in a range of fuel types.

Rain or dew quickly wets grassy fuel to the point where it will not burn. However, once rain has stopped the dead fuel moisture content of the grass will rapidly come back into equilibrium with atmospheric humidity. On bright sunny days in summer, grassy fuel will dry out to equilibrium moisture content within 2 h of even quite heavy showers.

3.5. Wind speed

Wind is the most dynamic variable influencing the behavior of grassfires. Wind speed varies widely over short time periods as well as with height above the ground (Finnigan, 2000). For the purposes of predicting rate of spread it is necessary to specify the height at which the wind is measured and the period over which it is averaged. The standard measure of wind speed for fire spread prediction is the value measured at 10 m above flat ground in the open averaged over 15–20 min.

Table 1 Likely dead fine fuel combustion characteristics for general bushfire fuels

Fuel moisture content value	Fine dead fuel combustion characteristics
>100%	Live green leaves—must be dried to burn
80%	Approximate wilting point—green leaves die and dry rapidly
30–35%	Fiber saturation level
28–30%	Eucalypt litter fuel will not burn
22–28%	Eucalypt litter fuel very difficult to ignite, burning difficult to sustain
20–24%	Grass fuel will not burn
16–22%	Eucalypt fuels difficult to ignite. Prescribed burning window. Pine fuels burn readily
13–16%	Litter fuel moderately easy to ignite, burning sustained. Pine fuels easy to ignite
10–13%	Burning in all fuels readily sustained, fire behavior predictable. Lower limit for low intensity prescribed burning
5–10%	Severe fire behavior, crowning in forests likely, progressively smaller firebrands start spot fires
3–5%	Extreme, erratic fire behavior, crowning, and spotting common

Provided grasslands are continuous, the rate of spread of fires is only about 20% lower in closely cropped or grazed pastures than in tall, undisturbed/natural grasslands. However, a fire in tall grass will have tall flames that may burn across narrow tracks, roads, or firebreaks that would stop a fire with lower flame heights in grazed pastures. When wind speeds are high, firebrands and embers can spread fire across quite wide firebreaks and the average spread across the landscape will approach the potential rate of spread in continuous fuels. At lower wind speeds (<20 km h^{-1}) a fire's average rate of spread across the landscape is likely to be less than the potential rate of spread in continuous grasslands.

When the wind speed at 10 m is less than 5 km h^{-1}, fires in continuous grass fuels will not spread in a consistent manner due to the variable direction of the wind. Instead, the direction of fire spread will be erratic as the fire responds to gusts in the flow. This wind speed is called the threshold wind speed (Cheney *et al.*, 1998).

The relationship between rate of fire spread in continuous fuel and mean wind speed greater than the threshold of 5 km h^{-1} for the three classes of fuel condition are shown in Fig. 5. The functions (Cheney *et al.*, 1998) are given as:

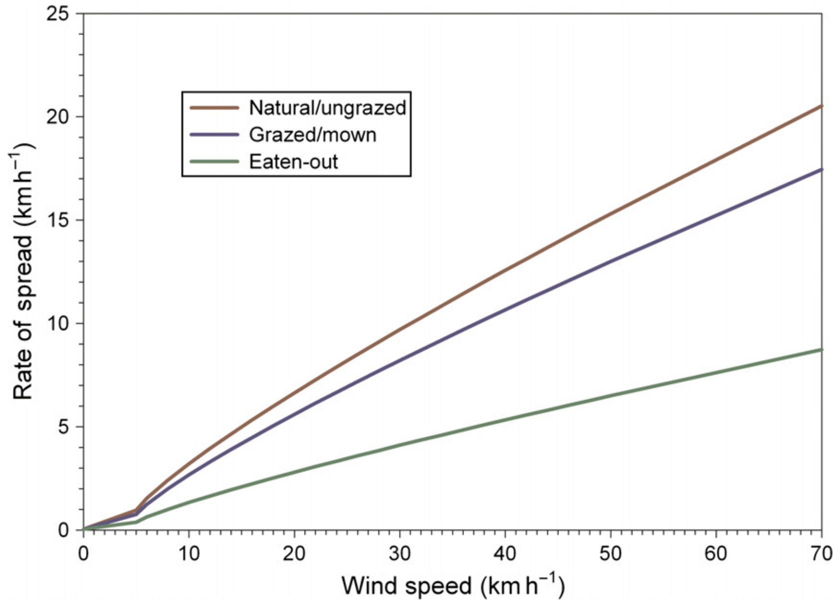


Figure 5 Wind speed functions for the three pasture conditions—natural, grazed, and continuous eaten-out. The rate of spread in natural pastures is approximately 20% greater than the rate of spread in grazed pastures.

Natural/ungrazed grass:

$$R = \phi M \phi C[1.4 + 0.838(U_{10} - 5)^{0.844}] \tag{4}$$

Grazed/mown grass:

$$R = \phi M \phi C[1.1 + 0.715(U_{10} - 5)^{0.844}] \tag{5}$$

Eaten-out continuous grass:

$$R = \phi M \phi C[0.55 + 0.357(U_{10} - 5)^{0.844}] \tag{6}$$

where R is the quasi-steady rate of forward spread (km h^{-1}), U_{10} is the average wind speed measured at 10 m in the open, ϕM is the fuel moisture coefficient given in Eq. (3), and ϕC is the curing coefficient given in Eq. (1). When the wind is below 5 km h^{-1}, the rate of spread is linear in U_{10}, ϕM, and ϕC. In fuels that are discontinuous, the threshold wind speed will be greater and dependent upon the degree of discontinuity.

Fires respond to both gusts and lulls in the wind, and both are incorporated in fire spread prediction through use of the average wind speed. Using instantaneous wind speed measurements or those averaged over a period shorter than 15–20 min will result in predictions that may

over- or underestimate the average spread rate of a fire depending on, if the measurement is made during a gust or lull. Similarly, making observations of fire behavior over short time periods may lead to erroneous estimates of the mean fire behavior and result in unsafe decisions being made.

3.6. Fire danger and rate of spread

The introduction of the CSIRO grassland fire behavior prediction system in 1997 brought about a significant change in the operational prediction of grassfire spread in Australia. The previous McArthur systems directly related the rate of spread of a fire to the danger (encompassing suppression difficulty) posed by the fuel and weather conditions in which it was burning. A survey of the fire authorities (Cheney *et al.*, 1990) found that while the calculation of rate of spread of McArthur systems may be deficient, the calculation of fire danger was still considered satisfactory. Thus, in revising the method to predict the rate of spread of a grassfire, Cheney and Gould (1995b) undertook to separate the calculation rate of spread from that of fire danger, electing to retain a slightly modified form of the McArthur Mk IV GFDM for fire danger.

The primary reason behind this separation was the identification of the need for a different wind speed function for different purposes. Figure 6

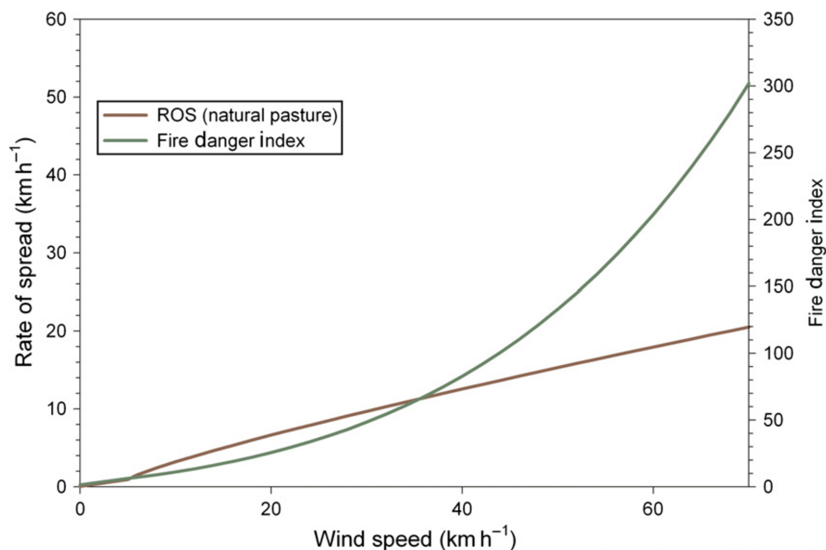


Figure 6 The wind speed function for rate of spread from the CSIRO Grassland Fire Spread Predictions System is a power law with an exponent less than 1. The wind speed function for fire danger from the CSIRO-modified McArthur Mk 4 Grassland Fire Danger Meter is an exponential that continues to increase with increasing wind speed.

shows the wind function of the Mk IV fire danger meter and that of the rate of spread in natural grass (Eq. (4)) for a given set of fuel and atmospheric conditions. The former is an exponential function in which the rate of increase of fire danger increases with increasing wind speed. By comparison, the rate of increase in the rate of spread function decreases with increasing wind speed (i.e., it is a power function with an exponent less than one). This function illustrates that with increasing wind speeds, the rate of spread of grassfires tapers off, whereas the fire danger continues to increase at an ever increasing rate, primarily due to the increased potential for fire ignition and increased difficulty of suppression.

The functions for effect of curing (Fig. 2) and dead fuel moisture content (Fig. 3) are also different but the wind function is the critical change.

3.7. Predicting rate of spread

Equations (1)–(6) form the CSIRO Grassland Fire Spread Prediction System and may be used to predict the rate of spread in all grass fuel types in Australia. They have been incorporated into software for point (CSIRO, 1996) and landscape fire spread prediction (e.g., Coleman and Sullivan, 1996). In order to facilitate prediction in the field, the equations were used to construct hand-held cardboard slide-rules, the GFSM (Fig. 7A), and the FSMNA (Fig. 7B).

Due to the logarithmic nature of the structure of the meters and the power-law function of the equations, incorporating more than one rate of spread function into a meter was not practical. As a result, a single rate of spread function for a particular pasture condition was used for each meter and the rate of spread for other pasture conditions approximated. In the case of the GFSM, which is intended for use in southern Australia where the standard summer fuel condition is improved pasture that is moderately grazed or mown, the base function is that for grazed pasture (Eq. (5)). The rate of spread in natural/ungrazed pasture or unharvested crops is then approximated from that of grazed or mown pasture using a factor of 1.2; the factor for rate of spread in eaten-out pasture is precisely 0.5 that of grazed.

In the case of the FSMNA, the base function is that for natural/ungrazed pasture. The rate of spread in woodland and open forest is then calculated using factors of 0.5 and 0.3, respectively, based on the measured wind reduction factor due to the presence of trees in these vegetation types.

The average, or quasi-steady, rate of spread should be taken as that measured over a period of 15–20 min; this accounts for short-term variations in spread caused by gusts and lulls in the wind and spatial variations in the fuel. The prediction system does not take into account the growth phase of a fire, and so will overestimate the spread early in the development of the fire or if the head fire remains narrow. Other factors in addition to the ones

described above, such as seasonal condition and fuel factors such as fuel fineness, were found to have a small influence on fire behavior and are incorporated into the system by way of definition, while others that also have a small effect (e.g., rainfall and solar radiation) were excluded.

In practice, a wildfire may burn over a variety of pasture types—including cereal crops and stubble fields, which may behave as either ungrazed or closely mown pastures—and rates of spread may vary accordingly from the values predicted using only one pasture type. It is recommended that predictions of fire spread in southern Australia be based on relationships for closely mown or cropped continuous pasture unless one of the other pasture

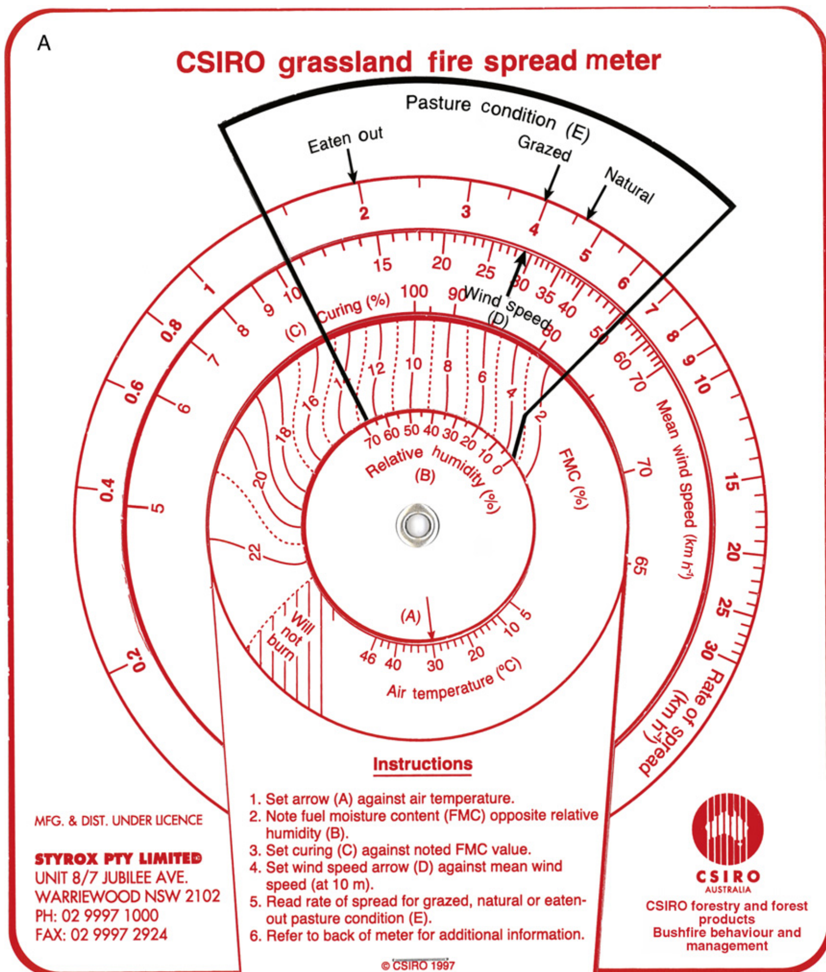


Figure 7 (continued)

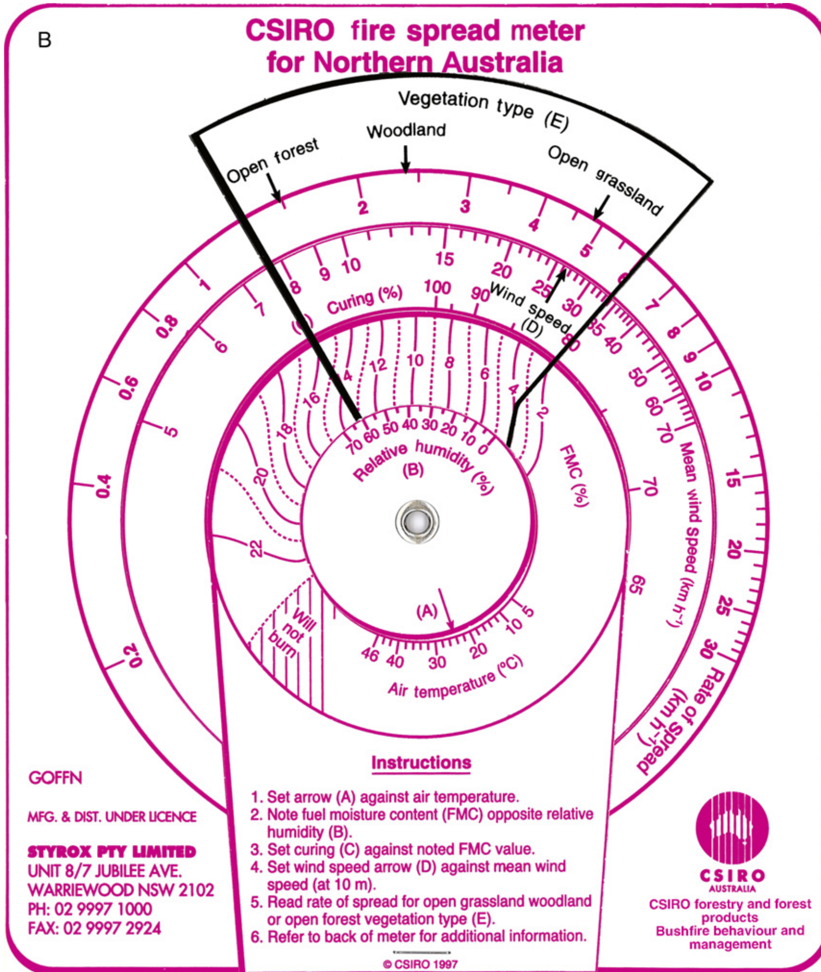


Figure 7 (A) The CSIRO Grassland Fire Spread Meter intended for southern Australia. (B) The Fire Spread Meter for northern Australia intended for tropical Australia.

conditions clearly dominates the landscape. For northern Australia, it is recommend that predictions of fire spread in open grassland, grassy woodland and open forest be based on the natural pasture condition. Differences in spread in these vegetation types are determined by differences in wind speed near the ground.

Although rain will obviously affect the behavior and spread of a fire, and in some circumstances extinguish it, it is not included as a factor in predicting grassfire spread because its effect on dead fuel moisture content is short-lived.

4. Impacts of Climate Change on Fire Spread Prediction

Recent studies of Australia's climate (Nicholls, 2003; Nicholls and Collins, 2006) have found that in general the continent has warmed by 0.85 °C since 1950 and the number of days with extremely high temperatures has increased. In the southeast, rainfall has decreased and droughts have become hotter. Climate change projections (e.g., Suppiah *et al.*, 2007) indicate that Victoria and NSW are likely to become hotter and drier in future, while Tasmania is likely to become warmer and wetter (McInnes *et al.*, 2004).

Hennessy *et al.* (2005) assessed the likely effect of forecast climate on the maximum daily forest and grassland fire danger indices for the years 2020 and 2050 using the daily extreme values for air temperature, relative humidity and wind speed as well as drought and soil dryness indices. The climate change forecasts, based on the change from present (1974–2003), for 2020 and 2050 were generated by two CSIRO global climate models (the Mark 2 and Mark 3 CSIRO Global Climate Models, GCM) using two emission scenarios based on the International Panel of Climate Change (IPCC) forecasts (IPCC, 2007). The first is a high emission scenario, with high climate sensitivity and the second is a low emission scenario, with low climate sensitivity. These scenarios included changes in average climate and daily weather variability and were applied to observed daily weather data. This method avoided biases often found when using raw climate model data that include changes in daily variability and inadequate assessment of changes in extreme weather events when applying changes in monthly average climate to observed daily data.

Hennessy *et al.* found that the sum of the average grassland FDI at 17 locations in southeastern Australia increased by around 0–15% by 2020 and by 5–30% using the Mark 2 CSIRO GCM (downscaled using the cubic-conformal atmosphere model, CCAM) and increased by 5–20% by 2020 and 15–40% by 2050, using the Mark 3 CSIRO GCM. Lucas *et al.* (2007) refined these results by including the latest IPCC climate scenarios and by using coinciding daily values for relative humidity and wind speed at 3 pm rather than the daily extremes for these variables. While the extrema of these variables may not occur simultaneously, relative humidity and dry air temperature are co-correlated and the argument that the daily maximum temperature occurs around 3 pm has some merit. Lucas *et al.* found similar results to Hennessy *et al.* with increasing occurrence of days of very high to extreme fire danger under the high emission scenarios.

While these forecasts provide some indication of the likely impact of climate change on the danger posed by bushfires in forests and grasslands under these scenarios and thus are extremely useful in assessing the likely

change in occurrence of fire weather, the prediction of fire danger tells us nothing about the likely impact of climate change on fire behavior, given the difference in the relationships of the independent variables described above (see Section 3.6). In this chapter, we will use a subset of the climate forecasts published by Lucas *et al.* (2007), namely the Mark 3 CSIRO GCM climate forecasts for the years 2020 and 2050 using only the high emission scenario (as it appears the low emissions scenario is no longer feasible), to estimate the change in the mean daily maximum (i.e., 3 pm) rate of spread of grassfires at three localities: Amberley (southeast Queensland), Canberra (Australian Capital Territory), and Adelaide (South Australia). These locations represent a cross section of fire seasons as defined by Luke and McArthur (1978) (Table 2), cover most of the primary climatic zones critical for the agricultural sector in Australia, and provide an indication of the likely widespread effect of climate change on the general fire behavior for these regions.

While fire behavior is a highly dynamic process that responds rapidly to changes in fuel and weather conditions, and the climate scenario data is currently limited to daily maximum values, the work of Lucas *et al.* to extend the initial climate data of Hennessy *et al.* to coincident 3 pm daily values does provide some insight into the likely scale of fire behavior for the day as this time has traditionally been used in the issuing of fire weather forecasts for the maximum temperature during the day and has some historical precedent. While wind speed is not satisfactorily represented in this manner, it is also not well documented in the historical dataset (Lucas *et al.*, 2007), nor is it forecast particularly well in the climate models (Lucas *et al.*, 2007). Thus, the results presented here provide only a preliminary indication of the likely change in fire behavior and are not conclusive. Improved forecasts of wind speed in climate forecast models will help improve this situation.

4.1. Climate change scenarios

The climate forecasts for the three selected regions based on the high emission climate change scenario are summarized by season in Table 3. A full discussion of the basis of these forecasts is given in Lucas *et al.* (2007).

Table 2 Fire seasons for locations as defined by Luke and McArthur (1978)

Location	Fire season
Amberley, Qld	(Aug) Sep–Dec (Feb)
Canberra, ACT	(Sep) Oct–Jan (Mar)
Adelaide, SA	(Nov) Dec–Feb (Apr)

Occasional early/late season extension months given in parentheses.

Table 3 Seasonal percentage changes of fire weather variables over present for the high emission climate change scenario

	Temperature		RH		Wind		Rain	
	2020	2050	2020	2050	2020	2050	2020	2050
Amberley								
Winter	105.4	115.6	99.3	100.2	98.6	98.0	100.1	105.3
Spring	104.7	113.7	98.3	96.4	101.0	103.6	98.2	92.9
Summer	103.8	111.1	99.0	98.6	100.3	100.9	99.7	99.1
Autumn	103.6	111.0	99.2	98.9	99.4	100.0	102.3	102.5
Canberra								
Winter	109.4	127.6	97.9	95.7	101.0	103.2	106.3	117.8
Spring	106.9	120.5	95.6	88.4	101.5	104.8	96.1	87.4
Summer	104.8	114.1	96.4	92.8	101.2	103.4	98.1	94.2
Autumn	104.3	112.8	98.7	97.6	101.8	105.4	104.4	111.9
Adelaide								
Winter	104.9	114.3	98.8	98.0	98.7	96.0	95.4	85.1
Spring	104.9	114.7	92.3	92.3	101.2	103.5	95.1	84.3
Summer	102.7	107.9	96.4	96.4	101.8	105.2	95.2	85.8
Autumn	103.0	109.8	98.8	98.8	100.3	101.8	98.1	95.4

100% represents current conditions (Source: Lucas *et al.*, 2007).

The original data consisted of daily 3 pm values for the four primary fire weather variables, air temperature, relative humidity, wind speed and rainfall, for the years 1971–2007 derived from historical records and equivalent datasets (i.e., 36 years) for each of the 2020 and 2050 climate scenarios. Thus the forecast datasets consist of 36 years of historical data to which the climate change scenario for each climate year (2020 or 2050) has been applied. The data are presented here as the mean seasonal percentage change from the current mean seasonal average and was obtained by averaging each value by Julian day across the 36-year period and segmenting into seasons.

Forecast mean daily 3 pm air temperature increases with time at all three locations, ranging from 3% to 16% at Amberley, 4% to 28% at Canberra, and 3% to 15% at Adelaide with the maximum increase occurring during Winter at Amberley and Canberra in both periods and Spring at Adelaide in 2050. Relative humidity generally decreases with time, with all sites experiencing the greatest decrease in Spring. Amberley experiences an increase in relative humidity in Winter in 2050. Wind varies the least of the four variables. Amberley shows a decrease in wind speed during Autumn and Winter in 2020 and a decrease only in Winter in 2050. The maximum increase in wind speed at Amberley of 3.6% occurs in 2050 during Spring. Canberra and Adelaide show increases in wind speed over all seasons, with

the exception of Winter in Adelaide, which decreases. The maximum increase of 5.4% occurs during Autumn at Canberra. The maximum increase in wind speed at Adelaide occurs during Summer. Forecast rain shows that Amberley and Canberra will have increased rainfall during Winter and Autumn, with a maximum increase of 17.8% at Canberra in 2050, and a decrease in rainfall during Spring and Summer at these sites and for all seasons at Adelaide. Adelaide shows the greatest decrease in rain (15.7%) during Spring in 2050.

4.2. Impact of changed weather on fire behavior

4.2.1. Method

Equations (4) and (5) were used to predict the mean daily rate of spread in natural and grazed grass fuels using the mean Julian day current and forecast weather data summarized in Table 3. Monthly mean values were then obtained for each month and the percentage change from current rate of spread for each fuel condition for each forecast year calculated. As with the fire danger modeling of Hennessy *et al.* (2005) and Lucas *et al.* (2007), it was assumed that grassland curing was 100%.

Strictly speaking, however, this assumption is only valid after grasses have flowered, set seed and senesced (Cheney and Sullivan, 2008), which depends upon the growing season (a function of species type, rainfall, and other weather factors). For most regions it will occur from mid to late fire season. It was assumed for the purposes of this chapter that for each location full curing, and thus the potential for widespread fire, would not occur prior to the onset of the normal fire season (i.e., September for Amberley, October for Canberra and December for Adelaide). Thus, predictions of the rate of spread are made only for each location's normal season as well as the extended late fire season as given in Table 2.

4.2.2. Results

Figure 8 shows the predictions of mean daily (at 3 pm) rate of spread in natural and grazed grass fuels for the three locations for the current historical fire weather and the forecast climate change-affected fire weather at 2020 and 2050, for the periods of each location's extended fire seasons. Also shown are the mean monthly 3 pm rates of spread for these combinations.

As expected, there is a general increase in the mean 3 pm rate of spread over the current mean values for the given climate scenario in 2020 and 2050, with the maximum increase occurring during late Spring and Summer at all locations. In the case of Amberley, there is little difference between the current rate of spread and the two forecast rates of spread in both fuel conditions late in the fire season. A similar reduction in difference occurs for Adelaide and Canberra in early Autumn. Table 4 summarizes the results on a seasonal basis in both natural and grazed pastures for each location.

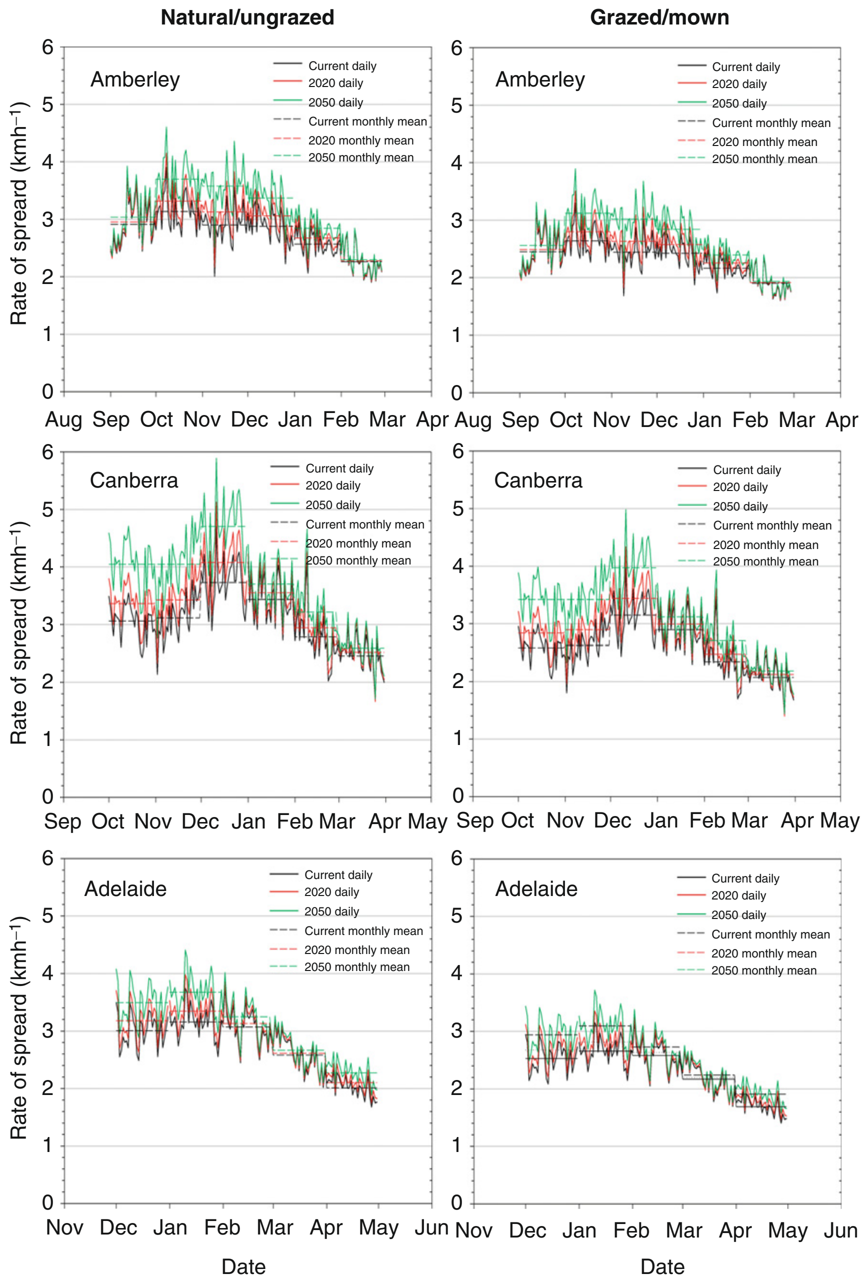


Figure 8 Predictions of fire behavior for current fire weather and four climate scenarios for three locations in Australia, from top to bottom: Amberley, Canberra and Adelaide. Left column is rate of spread in natural/ungrazed grasslands; right column is rate of spread in grazed/mown grasslands. Mean daily 3 pm values are given as solid lines, mean monthly 3 pm values are given as dashed lines.

Table 4 Mean monthly changes in rate of spread over current values in the three regions of interest (natural/grazed)

	Natural		Grazed	
	2020	2050	2020	2050
Amberley				
September	1.01	1.04	1.01	1.04
October	1.06	1.18	1.06	1.18
November	1.08	1.23	1.08	1.23
December	1.06	1.17	1.06	1.17
January	1.04	1.11	1.04	1.11
February	1.01	1.02	1.01	1.02
Canberra				
October	1.10	1.32	1.16	1.32
November	1.10	1.30	1.10	1.30
December	1.09	1.26	1.09	1.26
January	1.03	1.08	1.03	1.08
February	1.06	1.16	1.06	1.16
March	1.03	1.05	1.03	1.05
Adelaide				
December	1.06	1.16	1.06	1.16
January	1.06	1.16	1.06	1.16
February	1.02	1.06	1.02	1.06
March	1.01	1.03	1.01	1.03
April	1.05	1.13	1.05	1.13

The same relative change in the rate of spread of fire in natural and grazed fuel is due primarily to the fact that most of the change in the forecast climate is found in temperature and relative humidity, which is treated the same in both models in the fuel moisture content functions (Eqs. (2) and (3)). Where the models differ significantly, that is in the wind speed functions (Eqs. (4) and (5)), there is less relative change in the forecast values, thus little change in each rate of spread is noted. As a result, the remainder of this section and the following section refers only to the rate of spread in natural/ungrazed fuels.

In each case, the early fire season appears to show as much increase as the mid-fire season, although it is not as apparent at Amberley. The maximum increase in rate of spread occurs early in the fire season for Canberra (10% and 32% in October in 2020 and 2050, respectively) and Adelaide (6% and 16% in December). However, Canberra shows significant increase (over fire season mean of 19.5%) for the first 3 months of its fire season (32%, 30%, and 26%) in 2050, whereas Adelaide has a significant increase (over fire season mean of 10.8%) only for the first 2 months of its fire season (16% for both months) in 2050. By contrast, Amberley shows its biggest increase in the middle of its fire

season (8% and 23%) in November with the mid-fire season months showing the most significant increases (18%, 23%, and 17%) in 2050.

All locations show relatively little increase in rates of spread in the latter parts of the fire season (in the order of 1–6% by 2020 and 2–16% by 2050. Amberley shows the least increase in late season whereas Canberra shows the most.

4.3. Discussion

The effect of decreased rainfall in Spring experienced in all three regions is likely to decrease the growth rates and thus amounts of grass fuel present in the landscape, although this may be moderated somewhat by the increase in rainfall over Winter, particularly in southern Australia. No attempt has been made to predict the amount and coverage of fuel as fuel load does not play a role in determining rate of spread, although it may play a role in determining fire danger and difficulty of suppression, particularly in regions where the presence of increased fuel changes the likelihood of fire ignitions. Similarly, no attempt has been made to predict the onset of curing early in the fire season.

The onset of full curing early in the fire season in annual grasses is dependent upon the completed life cycle of each grass species and this in turn is dependent upon rainfall during the growth phase of the grasses. At all locations, Winter rainfall is expected to increase, by as much as 18% in southeastern Australia, which will provide an excellent basis for good growth. Now, while Spring rainfall is expected to decrease significantly in most regions, the effect of drought on grass curing will only be significant if this occurs late in the life cycle (i.e., after flowering and setting of seed). If regular, though diminished, rain continues to fall through to late Spring, then the curing process may be extended into early Summer and rates of spread will be reduced from those suggested here. On the other hand, once a grassland has flowered and set seed, curing may be accelerated by as much as a week by a single day of hot, dry, windy weather (McArthur, 1966).

As a result, it is possible that extensive fires may occur prior to the months of the fire season presented here. The method of determining the Julian day average for temperature, relative humidity and wind speed is not amenable to studying the effect of rainfall on the onset of curing and another method would have to be developed to incorporate this, for example, through ensemble modeling of rainfall occurrence across the years and the use of a pasture productivity model.

Regardless of when the onset of complete curing occurs, the increased rates of spread expected during the majority of the fire season for these locations means that there will be increased potential for large, conflagration-type fires that spread long distances across the landscape given sufficient levels of contiguous fuel. The impact of increased rates of spread is particularly important in the early phases of fire growth and may result in more rapid growth of fire size and thus, in turn, increase the rate of spread of a fire even further.

5. Implications for Grassland Fire Management

While the results presented in this chapter are based on fire weather variables that, firstly, are only the mean 3 pm daily value and not the daily mean, and secondly, whose variation may not be accurately reflected in climate models, they do represent an indication of the likely changes in fire behavior due to the impact of emissions on climate. As was suggested by the results of Hennessy *et al.* (2005), Lucas *et al.* (2007), and Pitman *et al.* (2007), there will be an increase in the daily mean 3 pm rate of spread in all grass fuels. However, while this increase is in excess of 30% for 2 months of the fire season in Canberra, it varies seasonally and in most months is not as significant a relative increase as that of fire danger.

Understanding how these increases in rate of spread due to changed climate will affect the day-to-day management in agricultural and pastoral lands is the key step in developing suitable adaptation pathways for land managers. As suggested by both the climate modeling and the fire behavior modeling results given above, this will depend upon the location in question. In southeastern Queensland, the rate of spread does not change substantially from current conditions late in the fire season in late Summer and early Autumn. Similarly, rate of spread in Canberra also drops off late in the fire season in Autumn. Adelaide experiences a similar reduction in effect about two-thirds of the way through the fire season from late Summer, although there appears to be a late season increase in April.

By the onset of the fire season in most regions, Winter cropping will have been harvested, leaving stubble and harvest residue unless continuous cropping has been implemented. Where stubble grazing is not employed, cereal crops may leave as much as 2–4 t ha^{-1} (Tolhurst *et al.*, 2008a), depending on the growing season and crop yield. If grazing is employed this may reduce to 1–3 t ha^{-1}. These fuels are, by their nature, discontinuous due to the planting of crop seed in rows and thus fires will not spread continuously in them in the absence of strong winds. Improved pastures for grazing are generally not sown in rows and thus will represent more continuous fuels and the potential for fire to spread continuously under lower wind speeds. Parrott and Donald (1970) found that while the above average Autumn rain (or irrigation) resulted in an increase of Winter production of annual pasture species, delayed desiccation in the Spring, and resulted in a greater yield of dry matter in the Summer late season, increased Spring rain (or irrigation) did little to prevent the maturation and desiccation of the annual plants. Thus, good Autumn rain and Winter growth will result in greater fuel loads. Followed by dry Spring and Summer seasons this will result in considerable potential for widespread fire.

Conservation tillage practices, such as direct drilling, minimum tillage or stubble retention, where the stubble residue is not plowed in or burnt off (or only lightly burnt), is likely to raise the overall equilibrium fuel load by 40% after 4 years compared to complete removal of harvest residue each year (Tolhurst *et al*., 2008a). While this additional residual fuel will be flat on the ground and may not play a role in the behavior of any fire, it may increase the continuity of the fuel layer and allow fire to spread in lower winds.

The likely increases in grassland fire behavior for the climate change scenario presented here will have a number of implications for both rural farm management and for fire management. The increased potential for larger fires resulting from increased rates of spread will mean that a number of changes in the current and evolving practices will be required.

The initial step in any fire mitigation endeavor is to stop fires from breaking out. Current systems differ slightly in implementation across the country but all are based on a forecast estimate of fire danger by the Australian Bureau of Meteorology using the CSIRO-modified McArthur Mk 4 Grassland Fire Danger Meter in the form of equations of Noble *et al.* (1980). Various levels of fire danger are used to restrict activities that may lead to the ignition of a fire. Generally, a FDI 50, all States enforce a Total Fire Ban in which no flames are allowed in the open. Other measures, such as closing public access to State forests or stopping high risk activities, are also often enabled. In South Australia, harvesting is prohibited once the FDI reaches 32. Use of grinders and other machinery that might cause a spark are also often prohibited under particular FDI values. With the increased risk of large fires early in the fire season there may be need to restrict high risk activities even further. The most likely mechanism for this would be to reduce the value of the FDI at which these prohibitions come into force. This may force rural landholders to conduct operations such as harvesting earlier in the morning or later at night in order for such prohibitions not to impact on the day-to-day conduction of their business.

The recent shift away from postharvest grazing of sheep toward continuous sowing (ABS, 2008) has led to a reduced need for fencing in order to facilitate greater cropping areas (Tolhurst *et al*., 2008a). The corollary is that with reduced need for fencing, there will be an inclination for less fire breaks (bare earth breaks in the vegetation designed to halt the spread of fires) as these use up valuable cropping area. While firebreaks are less effective under extreme fire weather conditions due to breaching of breaks by flame and transport of burning firebrands (Wilson, 1988), they do inhibit the growth (and thus width) of large fires, resulting in a reduced rate of spread (Cheney and Sullivan, 2008), and may halt the spread of small fires. Ensuring that a suitable network of firebreaks is maintained throughout the agricultural and pastoral regions will be essential to reducing the potential for large conflagration fires in the future.

Hand in hand with the necessary efforts to reduce the risk of ignition of fire and the preparation of firebreaks is the necessary effort to limit the spread of fires once they do break out. The key to any suppression action is early detection. A suitable, well-maintained network of fire towers across the agricultural zone and manned by well-trained staff during the fire season and, depending on the extent of Spring rain, perhaps earlier, is essential to aid in the early detection of fires. The size of farmholds and the low population density in many rural areas in the past has meant that a fire may be burning for some time before it becomes large enough to gain notice. The shift in ownership of many farms to corporately managed holdings means that often farms are frequently not occupied after harvesting has been completed, particularly if grazing is not employed, decreasing even further the likelihood of a fire outbreak being observed early.

Automated and remote methods for detecting the outbreak of bushfires have been under development for some time (San-Miguel-Ayanz and Ravail, 2005) and may provide additional capability to prevent the spread of large fires under future climate conditions. These methods include automated fire towers using visual and thermal cameras (Günay *et al.*, 2009; Martinez-de Dios *et al.*, 2008), airborne visual (i.e., spotter plane) and airborne hyperspectral detection techniques (Dennison and Roberts, 2009), and satellite detection using infrared (Pozo *et al.*, 1997), multiband and hyperspectral (Wang *et al.*, 2008) and techniques and combinations thereof (Giglio *et al.*, 2008), from a variety of platforms including AVHRR (Kant *et al.*, 2000), LANDSAT (Kant and Badarinath, 2002), and MODIS (Wang *et al.*, 2007). Networks of autonomous wireless inexpensive and disposable sensors deployed in the field (Kremens *et al.*, 2003) may also be of benefit in the early detection of fire outbreaks. However, no system is perfect and each has limitations in its timeliness, accuracy, and applicability (Briz *et al.*, 2003; San-Miguel-Ayanz and Ravail, 2005). Satellite-based detection methods are particularly limited in early detection due to the relative infrequency of the passage of many satellites over Australia. Effort must also be expended to ensure that false alarms are minimized in order to not diminish its utility.

Rapid deployment of suppression resources to a fire, once it has been detected, will increase the probability of containing the fire early (Plucinski *et al.*, 2007). With fast-moving, intense grassfires, direct attack of the headfire by ground crews will generally not be successful (McArthur, 1966). Instead, rapid response and suppression strategies designed to limit the growth in width of the fire will be critical to restricting the rate of spread and the final area burnt. By limiting the lateral spread through direct attack of the flanks, working forward from an anchor point at the rear of the fire, suppression crews will increase the chance of the fire being held by a firebreak. However, care must be taken when placing crews on flanks that might be subject to significant changes

in wind direction that will turn them into headfires. Aerial suppression has been found to aid in initial attack of bushfires (Plucinski *et al.*, 2007).

6. Conclusion

Agricultural farmland is an important resource in Australia, providing over 2% of the national gross domestic product and a significant source of exports, and provide the livelihood for many people in rural Australia. Protecting this resource is essential to Australia's economy and food security, particularly in the future as climate change due to global warming is expected to alter the weather, precipitating changes in many aspects of agriculture. Bushfires are a natural and essential part of the Australian landscape. Understanding the behavior of fires in natural and improved grasslands and the factors that influence that behavior is necessary to both control the occurrence of wildfires and implement fire as a strategic and tactical management tool. Understanding how changed climate will affect the behavior of grassfires will be critical to building adaptive strategies to manage fires in the landscape.

This chapter discussed the CSIRO grassland fire spread prediction system which was designed for both southern Australia, dominated by agricultural farmland, and northern Australia, dominated by native grasslands. The system, in the form of the CSIRO GFSM, has been in use by all state rural fire authorities in Australia since its introduction in 1997. The key independent variables in the system are mean wind speed measured at 10 m in the open on flat ground, the degree of grass curing of the grass sward, and the moisture content of dead fine fuel. Dead fuel moisture content is determined from the atmospheric relative humidity and temperature measured at 1.2 m in the shade.

The CSIRO GFSM was used in conjunction with a published set of climate change scenarios for the years 2020 and 2050 to determine the likely impact such changes in future fire weather will have on the rate of spread of grassfires in both natural and grazed pasture conditions at three locations in southeastern Australia: Amberley, Qld; Canberra, ACT; and Adelaide, SA. Modeling used the mean Julian day 3 pm values of wind speed, temperature, and relative humidity over each location's fire season and assumed grass curing of 100% to predict the mean Julian day rate of spread. These were then averaged over each calendar month to see the percentage change over current conditions.

It was found that the mean monthly 3 pm rate of spread during the peak fire season in natural pasture will be expected to increase by between 1% and 8% in 2020 and 4% and 23% in 2050 at Amberley; between 3% and 10% in 2020 and 8% and 32% in 2050 at Canberra; and between 2% and 6% in 2020

and 6% and 16% in 2050 at Adelaide. Similar increases were observed in grazed pastures. The greatest increases occurred in the early to mid-fire season at each location. The implications for management of grassland fires is that with the increase in rate of spread that grassfires will develop faster and increase the potential for large conflagration-type fires that cover extensive distances quickly and threaten assets and lives many kilometers from their point of origin. Early detection, fast initial attack, and direct attack of flanks to restrict the growth of such fires will be imperative under these changed climate conditions.

The fire spread modeling presented here makes the assumption that the grasslands will be fully cured by the start of each fire season and thus may overpredict the rates of spread early in the fire season. This will be particularly problematic with fire seasons following significant prefire season rain. The prediction of the expected rate of spread may be improved by undertaking modeling of the lifecycle of annual pastures to better predict the likely onset of curing at the beginning of each fire season.

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